UNLISTED SPECIES:

<u>Bank swallow</u>

Species description

The bank swallow is a small swallow with a dark back and white underparts with a dark breast band.

Life history

<u>Definition of suitable habitat</u>

The bank swallow breeds in lowland country with appropriate soft banks or bluffs and requires soft soil or fine sand for digging nest burrows. It generally avoids developed or forested areas and requires fresh, steep banks. Typical nest sites are found along low-gradient rivers (USDA Forest Service 1994). Preferred habitats for feeding include annual and perennial grasslands and lacustrine and riverine habitats (USDA Forest Service 1994).

<u>Reproduction</u>

Bank swallows typically nest in colonies, which can involve up to several hundred pairs of birds or thousands of burrows. They dig burrows along river banks, in sandy soils and clay banks (USDA Forest Service 1994). Birds may begin to arrive in breeding areas as early as late March and begin to disperse in late August (CDFG 1992).

<u>Diet</u>

These birds are completely insectivorous and feed by hawking insects in flight. They tend to be solitary feeders (USDA Forest Service 1994).

<u>Cover requirements</u>

Refer to the **Definition of suitable habitat** above.

<u>Dispersal</u>

Most yearlings return to the same colony or a nearby colony to breed. Site fidelity increases with age and past breeding success, with adults showing a strong tendency to return to the previous year's nest site (USDA Forest Service 1994). Breeding habitat may not persist for long periods in a single location, however, suggesting an ability to colonize new areas.

<u>Special habitat needs</u>

Banks with friable soils suitable for construction of burrows are essential to reproduction.

Current legal status

<u>Listing history</u>

The bank swallow is not currently a Federal candidate, proposed, or listed species. This species was listed as a threatened species by the State in 1989.

<u>Threats</u>

The channelization of rivers, erosion-control efforts, and bank stabilization projects destroy existing colonies and potential nesting habitat. Disturbance to colonies can occur as a result of recreation or other management activities (Remsen 1978).

Conservation needs

Pre-project surveys to locate colonies should be conducted prior to any activity which may impact colonies. Nest substrate at known colonies should not be altered. Disturbance buffers should be placed around active colonies from April 1 through August 1 (CDFG 1995). Unoccupied but suitable nesting habitat should be maintained for possible future colonization. Neither Federal lands nor PALCO lands figure prominently in the conservation needs of this species because it is not known to occur on these lands in northwestern California.

Status and distribution

Species

Numbers

No data exist on the California (USDA Forest Service 1994) or overall range-wide population size.

Distribution

Bank swallows are summer visitors and breed throughout North America except in Arizona, Arkansas, Texas, Louisiana, Alabama, Mississippi, Georgia, Florida, North Carolina, and South Carolina. The breeding range for this species is greater than 494,200,000 acres, however, the distribution is patchy. Bank swallows winter in central and eastern Panama, and in South America (USDA Forest Service 1994).

Reproduction

Range-wide Breeding Bird Survey data for 1966 to 1996 indicate a statistically non-significant trend of -1.3 percent (Sauer et al. 1997). However, the species has been extirpated as a breeder in many areas such as in southern California (CDFG 1992, Remsen 1978).

Suitable habitat

Acreage, distribution, and quality

No data exist on the amount or quality of suitable habitat range-wide. Within the range, habitat distribution is patchy and localized (USDA Forest Service 1994).

Pacific fisher

Species description

The fisher is a medium-sized carnivorous mammal, and a member of the family Mustelidae. Fishers have dark brown fur, and a general body build of a large stocky weasel. Adult males generally weigh between 7.7 and 12.1 pounds and are between 35 and 47 inches long. Adult females weigh between 4.4 and 5.5 pounds and are between 30 and 37 inches long (Powell 1993). Additional information on the description of the species can be found in Strickland et al. (1982).

The Pacific fisher is one of three subspecies in North America recognized by Goldman (1935) and Hall (1981). Hagmeier (1959) and Powell and Zielinski (1994), however, questioned the validity of these subspecies. Recent surveys indicate disjunct populations in the western states, therefore this analysis will treat the fisher populations on the west coast as a separate subspecies, and analyze impacts only to the western subspecies (Pacific fisher).

Life history

Definition of suitable habitat

Fishers are typically found in landscapes dominated by older successional stages of coniferous forest, and they use riparian areas disproportionally more than their occurrence (Powell and Zielinski 1994). Buck et al. (1983, 1994) noted that this subspecies appears to avoid areas with low (less than 40 percent) canopy closure. In northern California, however, fishers have been detected in open areas and in second growth-forests (Higley 1993, Self and Kerns 1995). Fisher use of these atypical areas is generally attributed to individuals foraging where prey availability may be higher.

Characteristics at resting and den sites are usually associated with older forests. Rest sites used by fishers include the canopies and cavities in large trees and snags, large down logs, deformities such as "witches brooms," and old squirrel and raptor nests (Powell and Zielinski 1994). In five studies in California, mean dbh for live trees and snags used for resting ranged from 24 to 46 inches (Buck et al. 1983, Seglund 1995, Zielinski 1995, Zielinski and Barrett 1995, Higley 1998). The Hoopa study found that live hardwoods ranging from 17 to 33 inches dbh were used most often (Higley 1998). Seglund (1995) reported the mean diameter for logs used for rest sites was 34 inches. Of 15 natal and maternal dens found in live trees, dbh ranged from 21 to 54 inches (Buck et al. 1983, Seglund 1995, Zielinski 1995, Zielinski and Barrett 1995, Higley 1998). Of five dens found in snags, dbh ranged from 29 to 58 inches (Buck et al. 1983, Seglund 1995, Zielinski 1995, Zielinski and Barrett 1995). Of the two maternal dens found in down logs on the Six Rivers National Forest, one was in a 39 inch (maximum diameter) white fir (*Abies concolor*) log, and the other was in a 79 inch (maximum diameter) sugar pine (*Pinus lambertiana*) log (G.A. Schmidt, pers. comm., January 8, 1999).

Fishers are sensitive to forest fragmentation (Rosenberg and Raphael 1986). In northern California, optimum habitat was reported to be comprised of 60 to 80 percent mature coniferous forest, 20 to 30 percent young mixed conifer and hardwood forest, and 2 to 5 percent pole-sapling forest (Buck et al. 1983). Klug (1996) found that fishers used redwood stands significantly less than Douglas-fir stands, and Douglas-fir stands used by fishers had significant hardwood components. Mast-producing hardwood species may be important providers of food to potential prey species (Klug 1996). In addition to possibly having a smaller prey base, Klug (pers. comm., January 15, 1999) hypothesized that redwood habitats also have fewer rest and den opportunities because fewer cavities and other deformities occur in redwoods relative to other tree species. Approximately one-half of the nine den sites found by Klug (pers. comm., January 15, 1999) in coastal northwestern California were in hardwood trees. Based on this information, Higley (1998) and Klug (pers. comm., January 15, 1999) concluded that the retention of large hardwoods is important in some habitat types for rest and den sites. Klug (1996) also speculates that hardwoods probably provides mast for fisher prey species (Klug 1996).

Home range sizes for fishers can vary substantially and are related to site-specific conditions such as topography, prey species diversity and density, and distribution of suitable rest and den sites. Using telemetry locations and minimum convex polygon methods, mean home ranges for male and female fishers in four California studies were 6,693 and 1,274 acres, respectively (Buck et al. 1983, Self and Kerns 1995, Zielinski 1995, Zielinski and Barrett 1997). These study areas were located in mixed conifer and Douglas-fir habitat types.

Reproduction

Females first breed at one year of age. Breeding typically takes place in March and early April, with young born the following year in early February or March. Fishers have a reproductive adaptation referred to as delayed implantation. This adaptation allows fishers to retain within the uterus a fertilized egg that becomes inactive for several months, allowing them to breed in early spring and not bear young until the following year. Females probably breed again within 10 days of giving birth. Fishers produce an average of three young per litter. Females use one to three dens per litter and often move kits from natal to maternal dens at 8 to 10 weeks. The female and young separate sometime between late summer and the first snows, with the males taking no part in caring for the young (Powell and Zielinski 1994, Strickland et al. 1982).

Diet

Fishers are known to eat small to medium-sized mammals, birds, and carrion (reviewed by Powell 1993). Fecal material collected at den and rest sites in southwest Oregon included remains of porcupine (*Erethizon dorsatum*), snowshoe hare (*Lepus americanus*), California ground squirrel (*Citellus beecheyi*), Douglas squirrel (*Tamiascurus douglasii*), northern flying squirrel (*Glaucomys sabrinus*), black-tailed deer (*Odocoileus hemionus*), pileated woodpecker (*Dryocopus pileatus*), hairy woodpecker (*Picoides villosus*), northern flicker (*Colaptes auratus*), and ruffed grouse (*Bonasa umbellus*) (Aubry et al. 1997). Stomach contents of eight carcasses from Trinity County, California, included false truffle (*Rhizopogon* sp.), bovine, brush rabbit (*Sylviagus bachmani*), black-tailed deer, broad-handed mole (*Scapanus latimanus*), and western gray squirrel (*Sciurus griseus*). While no porcupine remains were found in these carcasses, two of the specimens had quills embedded in their hides (Grenfell and Fasenfest 1979). Woodrats (*Neotoma* sp.) have also been detected in fisher scat collected in redwood types in northwestern California (R. Klug, pers. comm., January 15, 1999).

Cover requirements

A majority of the studies conducted in northern California show a preference of coniferous forests with high canopy cover. Carroll (1997), conducted a landscape-level spacial analysis and found fisher detections highly correlated to canopy cover and tree size. Refer to **Definition of suitable habitat** above and **Dispersal** below for additional information.

Dispersal

While independent from females by fall, young do not disperse from their mothers' home ranges until mid- or late winter. Fishers are thought to use riparian corridors and forested saddles between drainages for dispersal (Buck et al. 1983, Powell and Zielinski 1994). An aversion to open areas may limit population expansion and colonization of unoccupied habitat (Powell and Zielinski 1994). Klug (1996) suggested that due to the rapid growth of vegetation in the redwood region of northern California, 3 to 5 years may be enough time to allow sufficient regeneration in harvested stands so they no longer constitute a barrier to fisher movements.

Special habitat needs

Refer to **Definition of suitable habitat** above for specific habitat components needed for rest and den sites.

Current legal status

Listing history

The fisher is not currently a Federal candidate, proposed, or listed species. A petition to list the Pacific fisher as Federally endangered in California, Oregon, and Washington was found not to be warranted because substantial information supporting the requested action was not presented (USDI Fish and Wildlife Service 1991). A negative finding on an additional petition to list the fisher as Federally threatened in the western United States was made because substantial information indicating that Pacific Coast and northern Rocky Mountain populations constitute distinct vertebrate population segments was not presented (USDI Fish and Wildlife Service 1996b). The Pacific fisher is listed as endangered by the State of Washington, and as a species of special concern by the State of California

Threats

Historically, over-harvest (trapping) of fishers has resulted in population reductions and extirpations over much of their original range (Strickland et al. 1982, Aubry and Houston 1992, Powell and Zielinski 1994). Currently, the primary threat to the fisher is the reduction and fragmentation of late-successional forests, and the associated loss of habitat components necessary for resting and denning (Aubry and Houston 1992, Powell and Zielinski 1994). Increased fragmentation may cause fishers to travel long distances through unfamiliar or unsuitable habitat, thus increasing possible predation by coyotes, mountain lions, and other predators (Powell and Zelinski 1994). Based on the review of recent survey efforts, Pacific fisher populations may become increasingly genetically isolated throughout the western states. The apparent gap between populations can be as much as 500 to 600 miles long as in the case of the Southern Sierra population and the Klammath Mountains population in California.

Conservation needs

Additional research is needed on habitat use, food habits, and other aspects of fisher ecology (Powell and Zielinski 1994). Prohibition of legal harvest and efforts to reduce incidental trapping need to be continued (Powell and Zielinski 1994).

On Federal lands in Washington, Oregon, and northern California, the Northwest Forest Plan provides a network of large blocks of LSRs and interconnecting riparian reserves. Under this management regime the fisher was given a rating of 85 percent likelihood of viability, where it currently exists on Federal lands under the management of Northwest Forest Plan.

A majority of this subspecies range within the Sierra Nevada mountains is on Federal lands managed by the USFS. A regional management strategy to provide connectivity of late-seral habitat within the Sierra Nevada and between the Sierra Nevada and the Klammath mountain ranges is needed to provide for sustainable populations of fishers.

The coastal belt in California contains little Federal land. Historic trapping records indicate that Pacific fishers were not considered numerous in the coastal redwood belt of California, but few surveys have been conducted in recent years to determine the population status in this area. Based on historic data and indications from recent surveys (Beyer and Golightly 1996, Klug 1996) the role this coastal belt (including PALCO lands) plays in the conservation of the Pacific fisher is likely to be insignificant.

The amount and contiguity of late successional forests within the current and historical range needs to be improved, and important elements such as large trees, snags, and down logs need to be retained (Powell and Zielinski 1994). Habitat within the coastal redwood belt is likely to be low to moderate quality habitat for fishers. The following guidelines have been recommended for moderate quality habitat (Freel 1991, Heinemeyer and Jones 1994):

1. Maintain at least 40 percent of suitable habitat within a subdrainage as mature or older forests in patches of at least 80 acres in size.

2. Maintain riparian corridors at least 300 feet wide with at least 60 percent canopy closure.

3. Maintain 3 to 6 trees per acre with deformities or cavities that are at least 30 inches dbh.

4. Maintain 9 to18 live trees per acre in suitable habitat that are at least 20 inches dbh.

5. Maintain 1 to 2 snags per acre at least 30 inches dbh.

6. Maintain 2 to 3 snags per acre at least 20 inches dbh.

7. Maintain 2 to 3 down logs per acre at least 20 inches in diameter at the large end and 15 feet long.

8. Limit density of "open-to-public roads" to no more than 2 miles per square mile.

In addition to the guidelines suggested by Freel (1991) and Heinemeyer and Jones (1994), the following are additional guidelines based on review of recent fisher research conducted in California:

9. To provide adequate cover and potential foraging habitat for fishers, maintain at least 60 percent of each WAA in a vegetation seral stage classified under the CWHR system as CWHR 3M or larger (CWHR 3D, 4M, 4D, 5M, 5D, and 6) . Refer to Appendix 3 for a description of the CWHR vegetation classification system.

10. In areas where hardwoods are prevalent on the landscape, retain a hardwood component in the larger size classes present on the site.

Status and distribution

Species

Numbers

Based on trapping records and surveys most fisher populations are thought to have decreased range-wide since the late 1800s. There is no specific information available on the current number of fishers within the remaining areas occupied by fishers.

Distribution

In the 1800's, Pacific fishers were found in coniferous and coniferous-hardwood forests throughout Washington, Oregon, and California. The range of the Pacific fisher has contracted considerably from its original extent. In Washington, this subspecies only rarely occurs in the Cascade Range, Olympic Mountains, and in portions of the Okanogan Highlands (Aubry and Houston 1992). However, the Olympic Peninsula has been surveyed fairly extensively in more recent years, with no detections or confirmed sightings of fisher recorded (K. Aubry, pers. comm., November 5, 1998). They are also very rare in Oregon, although no thorough evaluation of their status and distribution has been conducted (Powell and Zielinski 1994). Limited surveys have been conducted in the coastal forests south of the Olympic Peninsula to Gold Beach, Oregon, but there have been no detections or confirmed sightings in this area (K. Aubry, pers. comm., November 5, 1998). Currently, Pacific fisher in California are only known to occur in the northwestern part of the state (northern Coast Range and Klamath Mountains) and in a disjunct population in the southern Sierra Nevada mountains of California (Zielinski et al. 1995).

The FEMAT (USDA Forest Service et al. 1993) lists the range of the Pacific fisher within the range of the northern spotted owl in Washington, Oregon, and northern California as covering 20,957,700 acres. This figure does not include the range of the fisher in the Sierra Nevada. A majority of this subspecies range is on Federal lands.

Reproduction
While the distribution of populations have decreased from historic levels, there are no specific data on range-wide Pacific fisher population trends.

Suitable habitat

Acreage
No precise estimate of the total amount of Pacific fisher habitat range-wide exists. Using northern spotted owl habitat combined with late-successional west-side Sierra Nevada mixed conifer and white fir as surrogates, there are an estimated 11,768,000 acres of Pacific fisher habitat in Washington, Oregon, and California (USDI Fish and Wildlife Service 1992a, Sierra Nevada Ecosystem Project Science Team 1996). Given the limited number of detections in several areas of the subspecies range, this is certainly an overestimate of the actual habitat occupied by this subspecies.

Distribution
Given the exceedingly low number of recent fisher detections in Oregon and Washington (Powell and Zielinski 1994), estimating the distribution of suitable fisher habitat in Oregon and Washington is difficult. No published descriptions of the distribution of fisher habitat in Oregon and Washington exist. Using spotted owl habitat as a surrogate for fisher habitat, more than 95 percent of the habitat in Oregon and Washington is found on federally managed lands (USDI Fish and Wildlife Service 1992a). In Oregon and Washington, suitable fisher habitat is probably discontinuously distributed throughout the Cascade Range, the Olympic Mountains, and the Coast Ranges.

Carroll (1997) mapped habitat suitability with a geographic information system (GIS) for fisher in northwestern California. He developed a multiple logistic regression model created by using data from survey locations and satellite imagery. Suitable habitat was predicted to be well distributed throughout northwestern California, including PALCO lands. This model predicted the largest concentrations of suitable habitat occurred on Federal lands administered by the USFS.

Late successional forests occur throughout the Sierra Nevada, however the major concentrations of high-quality late successional forest occur within National Parks and canyons of major river drainages along the western edges of National Forests (Sierra Nevada Ecosystem Project Science Team 1996).

Quality
Throughout their range, fishers display variation in habitat use. For example, in the eastern United States fishers occur in various age-classes of both hardwood and conifer forests, while in the Pacific States they appear to prefer late successional coniferous forests (Powell and Zielinski 1994). Not all habitats used, however, should be considered of equal quality without habitat-specific information that allows comparisons of survivorship and fecundity (Powell and Zielinski 1994). While coniferous LSH is generally considered suitable fisher habitat in the Pacific States, other habitats are undoubtedly of value to fishers as long as suitable canopy closure and specific

habitat elements (refer to **Definition of suitable habitat** above) are present. For example, Zielinski and Barrett (1997) found that Pacific fishers rested most frequently in stands classified as CWHR 4D, 5D, and 6. Zielinski and Barrett (1997) also found a presumed natal den in a stand classified a CWHR 3D. Using telemetry, Self and Kerns (1995) found fishers used CWHR 3D, 4D, 5P, and 5M stands disproportionally more than their availability, and that they avoided CWHR 3S and 4S. Klug (pers. comm., January 15, 1999) also thought that large expanses of LSH were not required by fisher, and that younger stands with residual old trees, LWD, and hardwoods would provide suitable habitat. Accordingly, the use of LSH to define fisher habitat should be considered conservative.

In the Pacific States, most fishers have been detected in low to mid-elevational forests up to 8,200 feet (Powell and Zielinski 1994). Low snow accumulation, and habitat characteristics that reduce snow depth such as high canopy closure, are thought to improve habitat quality (Powell and Zielinski 1994).

Fisher reaction to humans is one of avoidance; disturbance may cause fishers to move kits from dens (Powell and Zielinski 1994). Dark (1977) had more fisher detections in areas with low use roads than in areas of high use roads, and found that 83 percent of fisher locations were greater than 325 feet beyond human disturbance.

Within northern California, fishers are thought to be less common in coniferous forests dominated by redwood than they are further inland where Douglas-fir and hardwoods becomes more prevalent in coniferous forests (Beyer and Golightly 1996, Klug 1996). Early biologists also thought that Pacific fishers were rare in the redwood belt (Grinnell et al. 1937).

Red tree vole

Species description

The red tree vole is a small, microtine rodent. Females of the species tend to be slightly larger than males. Individual weight varies from 0.9 to 1.8 ounces (Hayes 1996). Its pelage is cinnamon to rusty brown in color.

Taxonomy: The red tree vole *(Arborimus longicaudus)*, formerly classified as *(Phenacomys longicaudus)*, is endemic to Oregon and possibly northern California. Its exact distribution is uncertain, but this species is believed to be restricted to mesic forest communities. The extent of its range in southern Oregon and northern California is in question. Until recently, the populations distributed throughout California and Oregon have been considered one species. In 1991, the California populations were proposed as a separate species, *Arborimus pomo* (Johnson and George 1991). The species *A . longicaudus* is believed to be isolated geographically and genetically from its sibling species *A. pomo* by the Klamath Mountains. However, recent DNA evidence suggests that the range for *A. longicaudus* may extend into Del Norte County in northern California (Murray 1995). This analysis will treat the populations of red tree voles in Oregon and California as two separate species, and analyze impacts on *A. pomo*.

Life history
Little site-specific information on habitat use or population numbers has been gathered within the range of *A. pomo*. The voles in California are found in different forest types than members of the same genus in Oregon. Since little work has been done on the habitat associations of the vole in California, much of what is known has been derived from studies in Oregon forest types. For the purpose of this analysis it is assumed that habitat use, population structure, and reproduction, of *A. pomo* is similar to that of *A. longicaudus*. Many of the references listed below are from study sites in Oregon.

<u>Definition of suitable habitat</u>
In California, red tree voles are most commonly found in coniferous stands which have a component of Douglas-fir, though they are also found in stands with grand fir (*Abies grandis*), Sitka spruce (*Picea sitchensis*), and western hemlock (*Tsuga heterophylla*) (Huff et al. 1992). The species is nocturnal, and voles may spend the majority of their lives in the forest canopy, moving from tree to tree through the canopy (Carey 1991). Although they are almost exclusively arboreal, some terrestrial activity does occur; and occasionally individuals have been captured on the ground (Corn and Bury 1991, Raphael 1988).

Red tree voles build conspicuous nests, predominately in Douglas-fir tress wherever there is a suitable foundation and readily accessible food supply (Carey 1991). These nests are inhabited year-round and provide shelter, protection from predators, and micro-climates suitable for rearing young. Nests are constructed of resin ducts of fir needles, lichen, feces, urine, conifer needles, and small twigs (Carey 1991). The resin ducts are definitive indicators of tree vole use of a nest structure. Multiple generations of voles may use the same nest, continually enlarging it; a large nest may have several chambers and tunnels.

Nest sizes are variable, ranging from fist-sized to as large as 3 feet in diameter. Nest size varies with the size and limb structure of the tree supporting the nest. Some nests are found on large single branches or whorls of branches, and some are against the bole. Biswell (1996, cited in Behan et al. 1996), working in the Oregon Coast Range, found that approximately 40 percent of nests were on single large branches. Single branches supporting nests averaged greater than 4 inches in diameter. In trees with very large branches, tree vole nests can be 10 to16 or more feet from the bole of the tree.

The home range size for this species is not well known and is likely to vary depending on habitat quality. However, Biswell (1996, cited in Behan et al. 1996) found individual adult red tree voles, radio-tracked for 35 to 106 days, used 2 to 7 (median = 5) nest trees having independent non-interconnected canopies. The greatest straight-line distance traveled between consecutively occupied nest trees was an overnight move of 248.7 feet. The mean distance moved between consecutive nest trees for males and females combined was 112.8 feet (SE = 21.3). When moving to a new nest tree, adult voles re-occupied previously constructed nest structures at least 68 percent of the time. Thirty-six percent of nest trees located via telemetry (n=39) contained more than a single nest, and one tree contained seven nest structures.

Reproduction
Reproduction in this species is characterized by a long reproductive period, small litter size, and slow development of young (Carey 1991). Red tree voles can breed throughout the year, but generally litters are produced from February through September (Carey 1991). Litters range in size from one to four (Carey 1991, Maser 1966), but average two (Howell 1926). Gestation is approximately 28 days but may extend to 48 days if the female is lactating in support of an earlier litter (Carey 1991). The species is believed to exhibit sexual segregation except for the purposes of reproduction. Adult males and females rarely occupy the same nest at the same time (Whitaker 1998).

Diet
Red tree voles feed primarily on Douglas-fir needles, although they will occasionally feed on grand fir, white fir, Sitka spruce, and western hemlock needles (Carey 1991). Douglas-fir needles have resin ducts along each edge which the vole discards, eating the fleshy portions of the needles. Water is obtained from dew, rain or condensation on foliage (Carey 1991).

Cover requirements
Biswell (1996, cited in Behan et al. 1996) found nest trees generally had independent, non-overlapping canopies requiring the voles to move on the ground between some nest trees. Old-growth habitat appears to provide optimum habitat for red tree voles because it functions as a climatic buffer and has a high water-holding capacity, which maximizes food availability and free water (Gillesberg and Carey 1991).

Dispersal
Young start to venture from the nest at about 4 weeks of age (Howell 1926). Eventually juveniles will leave the nest to establish their own nests. The greatest distance moved by a red tree vole was by a dispersing male that was followed for 40 days. This individual was located in five different trees and reached a maximum straight-line distance from his natal nest tree of 1,115 feet (Biswell 1996, cited in Behan et al. 1996). While moving far greater distances than adults, subadults have extremely low survival rates. Telemetered red tree voles crossed small forest roads, small streams, or canopy gaps while traveling between nest trees.

Special habitat needs
Information on the special habitat needs can be found under the **Diet** section above.

Current legal status
Listing history
The California red tree vole is not a Federal candidate, proposed, or listed species under the Act, and is classified by the CDFG as a mammal species of special concern. The red tree vole in Oregon (*A. longicaudus)* was designated as a "Survey and Manage Species" on Federal lands within the range of the northen spotted owl under the Northwest Forest Plan. Due to the

uncertainty of the taxonomy of red tree voles at the time the Northwest Forest Plan was developed, *A. pomo* was not listed as a "Survey and Manage Species".

<u>Threats</u>

A majority of the species' range is on private land which is intensively managed for timber production. Its apparent affinity for coastal old-growth and late-seral forests with a Douglas-fir component, has increased the level of concern for this species.

Huff et al. (1992) rated the red tree vole as the most vulnerable of the arboreal rodents to local extirpations resulting from the loss or fragmentation of old-growth Douglas-fir forests. The California red tree vole is classified by the CDFG as a mammal species of special concern (Williams 1986) because of its affinity for coastal conifer forests and the potential negative impact of large-scale timber harvest.

<u>Conservation needs</u>

The red tree vole is one of the least studied of the arboreal rodents occurring in Douglas-fir forests in the Pacific Northwest (Carey 1991). Most of what is known comes from anecdotal observations and a few limited studies. Little information is available on the major aspects of this species life history (longevity, demography, or population density). Additional research on the effects of landscape characteristics (seral stage mix and habitat fragmentation) and distribution and abundance of red tree vole populations is needed to develop a scientifically based conservation strategy. Based on the level of information available on the range-wide status and distribution of this species, the importance of the role PALCO lands play in the conservation of the California red tree vole is unknown.

Based on the research conducted to date on the California red tree vole and its close relative (*A. longicaudus*), the following should be considered when developing management guidelines for the conservation of the species:

1. Provide an interconnecting network of blocks of LSH and opportunities for dispersal of young and movement of adults between the LSH blocks.

a. Little information on the relationship of patch size and habitat suitability for red tree voles is available. Of the research conducted to date, information collected from the Oregon coast range is most similar to PALCO lands, in terms of habitat type. Based on stand sizes recorded in Huff et al. (1992), patches of suitable habitat should be a minimum of 75 acres in size, and preferably greater than 475 acres in size. These habitat patches should contain an element of older trees with a limb structure adequate to support large vole nests .
b. Red tree voles are thought to spend a majority of their life within the canopy of coniferous forests, moving from tree to tree through the canopy. To increase the likelihood of successful movement between colonies of red tree voles, dispersal habitat

between patches of suitable habitat should consist of coniferous forest with canopy cover of at least 60 percent.

Status and distribution

Species

Numbers and Distribution

The California red tree vole is endemic to California. Approximately 78 percent of the species range occurs on non-Federal lands (USDA Forest Service et al. 1993). There is little information available on the current population size or status of the California red tree vole. The CDFG is in the process of gathering information on the status and distribution of red tree voles in California. Their records indicate that the range extends from San Francisco Bay north along the coast and east as far as the Klammath Mountains in Sonoma, Mendocino, Trinity, Humboldt, and Del Norte Counties.

The CDFG has recorded approximately 561 red tree vole nest locations and 388 trapped individuals within the range of the California red tree vole. The sites in this database represent a small portion of the potentially suitable habitat within the species range. The CDFG database includes records dating back to 1984, with a majority of the sighting information collected since 1994. The distribution of locations currently in their database indicate that red tree voles are more numerous along the coast than inland, and that their distribution inland may be limited (CDFG 1997).

Based on the information contained in the CDFG database, the overall distribution of red tree voles has not changed significantly, although the apparent limited mobility and dispersal capability of the species is of concern. The continued decrease and fragmentation of late-seral habitat within the species range is likely to reduce population sizes and limit the species' ability to recolonize areas.

Reproduction

There is no information available on the reproductive trends of the California red tree vole. Surveys indicate that late-seral forests contain larger populations than younger stands. Larger populations have a higher likelihood of persistence and may equate to higher levels of reproductive success.

Suitable habitat

Acreage

Acres of potentially suitable habitat within the range of the California red tree vole are discussed **in the Environmental Baseline (in the action area)** section of this document.

Distribution

No specific information is available on the distribution of potentially suitable red tree vole habitat. Based on CDFG (1997) sighting records of red tree voles and red tree vole nests, potentially suitable habitat exists in Sonoma, Mendocino, Del Norte, Humboldt, and Trinity Counties.

Quality
Little is known about the number or size of Douglas-fir trees, or the stand structural characteristics required to sustain a local population of red tree voles. Voles have been found in all seral stages of Douglas-fir forests from closed sapling/pole stands to older stands (Corn and Bury 1986) but tend to be significantly more abundant in mature and old-growth stands (Corn and Bury 1986, Aubry et al. 1991, Huff et al. 1992). Individuals were captured in clearcuts (Corn and Bury 1986), young second-growth Douglas-fir stands, and old-growth (Corn and Bury 1986, 1991, Gomez 1993, Gillesburg and Carey 1991, Zentner 1977). Capture rates were significantly higher in old-growth Douglas-fir forests than in young (40 to 60 year old) or natural mature forests (Gomez 1993, Corn and Bury 1991). In northern California, Zentner (1977), found old-growth Douglas-fir stands contained more red tree vole nests and larger colonies than did second-growth stands. The youngest stand in which red tree voles were captured in the Oregon Coast range was 62 years old (Huff et al. 1992). In Mendocino County, California, Meiselman (1996) found red tree vole nests significantly more abundant in old-growth stands (greater than 200 years) than in mature (100 to 200 years) or young stands (less than100 years), despite the fact that the difficulty in detecting nests in the upper canopy of old-growth stands may have resulted in an underestimate in old-growth forest.

In a random sample of stands in the central Coast Range of Oregon, red tree vole nest tree densities averaged 2.29 per acre (range 0.8 to 33.6 per acre) in 150 to 300 year old stands, and 0.16 per acre (range 0 to 4.8 per acre) in 25 to110 year old stands (Biswell 1996, cited in Behan et al. 1996).

The relationship between number of nests and number of individuals is largely unknown (Carey et al. 1991). As the size of a tree and its branches increase, the amount of suitable habitat within the tree also increases, making it more likely the tree will be used for nesting by red tree voles (Gillesberg and Carey 1991). The large limbs of old-growth trees provide the structural support for large nests, as well as escape routes. However, vole populations are often patchily distributed in forests.

The presence of Douglas-fir clearly is important to maintaining viable populations of red tree voles. Huff et al. (1992) found that even though basal area and density were highly variable among stands, the basal area of Douglas-fir was greater than 40 percent of the total stand basal area in 15 of the 18 stands where red tree voles were captured in Oregon. In the Oregon Coast and Cascades Ranges combined (Huff et al. 1992) found that stands with red tree voles had a mean of 12 large (greater than 39 inches dbh) Douglas-fir trees per acre, whereas those without voles had significantly fewer large Douglas-firs (6 per acre; P=0.02).

Red tree voles were captured in stands ranging in size from 75 to 1,280 acres (mean = 475 acres) in the Oregon Coast Range and were not captured in stands less than 75 acres in size (Huff et al. 1992). There is no conclusive information available concerning the minimum size stand necessary to support a population of red tree voles. Factors such as the number of suitable nest trees, canopy closure, and past and present disturbances may be more important to the suitability of a stand than its acreage.

<u>**Northern red-legged frog**</u>

Species Description

The northern red-legged frog (*Rana aurora aurora*) is a moderate sized, brown or reddish-brown frog usually marked with small black flecks and spots on the back and sides, with dark bands across the legs. A dark mask is generally present, with a light upper jaw strip extending nearly to the shoulder. They have smooth, moist skin, with eyes oriented to the sides. The ventral surface of the hind legs are reddish in color, often extending onto belly and sides (Leonard et al. 1993). Captive northern red-legged frog have been reported to live 12 to 15 years.

Two subspecies of red-legged frogs occur in California; the northern red-legged frog and the California red-legged frog (*Rana aurora draytonii*). While the genetic relationship between the two is unclear, some morphological differences are evident. Davidson (1996) reviewed literature on the two subspecies. The most obvious differences include the larger size of the California red-legged frog which calls above water and lays its eggs in contact with the water surface. The northern subspecies is smaller, the male typically calls under the water, and eggs are submerged.

Life History

<u>Definition of suitable habitat</u>

Key elements for the northern red-legged frog include both aquatic habitats for breeding and terrestrial habitats for foraging and overwintering. Breeding habitats include vegetation-lined marshes, bogs, swamps, ponds, lakes, and slow-moving streams. Outside of the breeding season, adults are highly terrestrial and are frequently encountered in forested stands adjacent to streams (Leonard et al. 1993). Nussbaum et al. (1983) reported finding northern red-legged frogs 650 to 975 feet from water. Zeiner et al. (1988) reported red-legged frogs found "considerable distances" from breeding sites during rainy periods.

The specific habitat parameters needed for overwintering and foraging in forested environments are not well understood, however some correlations have been made with coarse woody debris, canopy closure, and stream attributes within and adjacent to breeding sites. Along the west slope of the Cascades, Bury and Corn (1988) found northern red-legged frogs more abundant in mature and old-growth habitats compared to young stands or clear-cuts, but these authors indicate the proximity of slow-moving creeks or ponds may have influenced the presence of the subspecies in adjacent terrestrial habitat. The authors also surmised the location of northern red-legged frogs in young stands could be attributed to the presence of closed canopy and coarse woody debris within those stands. In southern Washington, Aubry and Hall (1991) found the subspecies most frequently in mature stands and least frequently in young stands, but reasoned that their presence may be more correlated with instream downed wood and pools than with stand age. In the coastal redwood zone, northern red-legged frogs are commonly found on stream side benches with dense sword fern undergrowth (Twedt 1993). There may be competitive advantages to inhabiting isolated perennial pools within intermittent stream reaches (Hayes and Jennings 1988) if water temperatures are suitable.

Reproduction

Males are known to arrive at breeding sites at least 1month prior to females, with water temperatures as low as 35.6 degrees Fahrenheit (Licht 1969, cited in Jennings and Hayes 1994). Breeding usually occurs in February and March. Eggs are laid in clutches of 700 to 4,000 (Zeiner et al. 1988) and are attached to underwater vegetation at least 2 to 3 feet from the water's edge, at a depth of at least 18 inches (Licht 1971). Egg development can vary from 1 week to over 8 weeks depending on temperature (Storm 1960 and Licht 1971, cited in Jennings and Hayes 1994). Lethal minimum and maximum embryonic temperatures are 39 and 70 degrees Fahrenheit respectively (Licht 1971). Aquatic larvae take up to 4 to 5 months to metamorphose, and are about 0.75 inches long when they become terrestrial. Nussbaum et al. (1983) refer to limited evidence from western Oregon that indicates sexual maturity is achieved in the second year following metamorphosis. Leonard et al. (1993) indicate 3 or 4 years likely are required for the subspecies to reach sexual maturity.

Diet

The diet of the red-legged frog species is reported to be highly variable (Hayes and Tennant 1985, cited in Davidson 1996). The northern subspecies specifically, is noted as being primarily insectivorous (Licht 1986, cited in Jennings and Hayes 1994). Jennings and Hayes (1994) note that northern red-legged frog tadpoles can significantly reduce the standing crop of epiphytic algae under certain conditions.

Cover requirements

Refer to Jennings and Hayes (1994) for a review of the published literature. In summary, the northern red-legged frog utilizes aquatic vegetation bordered by dense grassy or shrubby vegetation. In terrestrial areas, patches of vegetation such as willow thickets and dense sedge swales are used. Their escape strategy from birds and land predators is to flee directly into the water and swim to the deepest part of the channel or pool (Gregory 1979). As described by Twedt (1993), the northern red-legged frog in northwestern California has been found in dense undergrowth of sword ferns, and sedges.

Dispersal

Information on dispersal distances or specific dispersal habitat requirements was not available.

Special habitat needs

The presence of instream downed wood and pools at least 18 inches deep appear to be important for northern red-legged frogs. See **Definition of suitable habitat** above for additional information.

Current legal status

Listing history

The northern red-legged frog is not currently proposed or listed as Federally threatened or endangered. It is currently identified as a California species of special concern and is recognized by the USFS as a Sensitive Species in the Pacific Southwest Region. Red- legged frogs found from southern Del Norte to northern Marin county exhibit intergrade characteristics of both subspecies (Hayes and Krempels 1986, cited in USDI Fish and Wildlife Service 1996c). Northern Marin county is considered the approximate dividing line between the Federally listed California red-legged frog and the intergrade zone along the coastal range (Mark Jennings, pers. comm., 1993, cited in USDI Fish and Wildlife Service, 1996c). The project area occurs within the zone of intergrade of the two subspecies. The Final Rule for the listing of the California red-legged frog does not include the intergrade zone, and does not include the following areas: the State of Nevada, Humboldt, Del Norte, Trinity, and Mendocino counties, California; Glenn, Lake, and Sonoma counties west of the Central Valley Hydrological basin, California; or Sonoma and Marin counties north and west of the Napa River, Sonoma Creek, and Petaluma River drainages, which drain into San Francisco Bay, and north of the Walker Creek drainage, which drains into the Pacific Ocean, California.

Threats

All life stages are susceptible to competition and/or predation from introduced fish and bullfrogs (*Rana catesbiana*) (see Davidson (1996) for a review of the literature). The FWS has been unable to locate studies or data to indicate how serious the competition and predation may be, or what trend may be emerging in the forested portion of the species range. As with many other amphibian species, the northern red-legged frog is also vulnerable to the effects of cattle grazing due to the removal of protective stream-side vegetative cover and vulnerability to egg mass trampling. Additionally, habitat removal caused by timber harvest adjacent to or within riparian areas poses a threat to the subspecies since it is dependent on terrestrial vegetation for much of its life cycle. Welsh et al. (1998) suggest that aside from habitat destruction, the use of forest herbicides may be the single greatest threat to the northern red-legged frog in Humboldt County.

Conservation needs

Jennings and Hayes (1994) recommend systematic surveys of the northern red-legged frog are needed to further understand how serious impacts on this taxon are and what trends may be evolving. Welsh et al. (1998) recommend forested and riparian areas, including all standing water bodies, be managed as if they currently have the potential to support breeding northern red-legged frogs. These areas should be free of introduced predators. Wide buffer widths which restrict all habitat removal and alteration from riparian and adjacent forested areas would likely result in long-term maintenance of northern red-legged frog populations (Welsh et al. 1998). Potential impacts to northern red-legged frogs from herbicide and pesticide run-off should be investigated (Nussbaum et al.1983).

Status and Distribution

Species

Numbers

Range-wide population data are not available for the northern red-legged frog.

Distribution
The northern red-legged frog is found in streams in forested areas from Sullivan Bay, British Columbia, south along the Pacific coast (west of the Cascade range) to northern Humboldt county, CA (Stebbins 1985), or northern Del Norte county as described by USDI Fish and Wildlife Service (1996c). It interbreeds with the California subspecies along a 40 to 60 mile wide diagonal band that runs from southwestern Del Norte county to northern Marin county (range map, Stebbins 1985). The action area is within the interbreeding zone, where frogs exhibit primarily features associated with the northern red-legged frog (Jennings and Hayes 1994). The northern red-legged frog is generally limited to elevations below 3,936 feet (Zeiner et al. 1988).

Range-wide data on the northern red-legged frog were not summarized in the FEMAT (USDA Forest Service et al. 1993) analysis. However, range maps (e.g., Corkran and Thoms 1996) indicate the range of the northwestern salamander (*Ambystoma gracile*) (summarized in FEMAT) overlaps the northern red-legged frog range by 85 to 90 percent in Oregon, Washington and California. The noticeable difference is that the range of the northwestern salamander extends to higher elevations along the Cascade crest.

Suitable habitat
Acreage, distribution, and quality
Specific range-wide data on suitable habitat is not available for the northern red-legged frog. However, the range of the northwestern salamander is similar to the range of the northern red-legged frog. We can use the similarity of range with the northwestern salamander as described in FEMAT (USDA Forest Service et al. 1993) as a proxy for the northern red-legged frog. The FEMAT (USDA Forest Service et al. 1993) data for the northwestern salamander are as follows. The portion of the species range in the United States is approximately 37.18 million acres in size, of which approximately 38 percent (14.13 million acres) is Federal land. Federal lands in this species range consist of approximately 25 percent Matrix (3.5 million acres), 7 percent Adaptive Management Area (988,870 acres), and 68 percent reserved or withdrawn from management (9.6 million acres). The Northwest Forest Plan estimated 2.63 million acres of Riparian Reserves occurred within Matrix land allocations. Mapped wetlands were likely included in these figures. This number excludes the acres of Riparian Reserves in other land allocations, but based on a sample, an estimated 40 percent of other land allocations is comprised of Riparian Reserves on Federal land throughout the range of the northern spotted owl (Northwest Forest Plan Record of Decision, page B-12). Using this percentage, an estimated 5.65 million acres of Riparian Reserves occur within the range of the northwestern salamander, and for the purposes of this analysis, the northern red-legged frog. An unknown portion of this acreage is probably unsuitable, especially near high-gradient waters.

The following figures are estimated based on adaptations from FEMAT (USDA Forest Service et al. 1993) and describe miles of fish bearing (Class I) and non-fish bearing streams (Class II) on National Forests and BLM lands within the range of the subspecies: 16,400 miles of Class I and 32,860 miles of Class II occur within the range of the northern red-legged frog. The following physiographic provinces were included in the above estimate: Western Cascades, Western

Lowlands and Olympic Peninsula of Washington; Western Cascades, Coast Range and Klamath Range of Oregon; and the California Coast and Klamath Ranges.

Data on how much overlap of habitat occurs where streams converge is not available at this scale, and therefore overestimates potentially suitable habitat. Conversely, these figures do not consider potentially suitable habitat on private ownership, therefore, the acreages described above likely underestimate potentially suitable habitat for the northern red-legged frog.

Foothill yellow-legged frog

Species description

The foothill yellow-legged frog is a small to medium sized frog found in lower gradient, rocky and gravelly streams. It has grainy skin with colors ranging from olive to light or dark gray or brownish, with variable amounts of brick red. The underside is white or cream, with light or bright yellow along the sides of stomach and undersides of thighs. A comparative study of ranid frogs (Duelmann and Trueb 1986) indicates that the lifespan may be a dozen years or more.

Life history

Definition of suitable habitat

The foothill yellow-legged frog is closely associated with shallow areas of permanent streams and is most common in and near streams with rocky, gravelly, or sandy bottoms (Leonard et al. 1993). The species is also occasionally found in other riparian habitats including moderately vegetated backwaters, isolated pools, and slow moving rivers with mud substrate. No information was found on specific water temperature needs, and all life stages throughout the species range appear to occur in highly variable water temperatures. Cover immediately adjacent to the water's edge can be somewhat sparse and patchy (Ashton et al. 1998). The species occurs in many types of mediterranean ecosystems. Zeiner et al. (1988) identified the following California vegetation communities adjacent to streams as potentially suitable habitat for foothill yellow-legged frogs: valley-foothill hardwood, valley-foothill hardwood-conifer, valley-foothill riparian, ponderosa pine, mixed conifer, coastal scrub, mixed chaparral and wet meadow types. Little is known about the areas used for overwintering habitat. Foothill yellow-legged frogs are known to leave the stream channel during high flows and hibernate in the forest, sheltering in animal burrows and under logs. They have been captured up to 328 feet from the South Fork Eel River (Welsh et al.1998).

Reproduction

Courtship and breeding occurs on gravel, pebble and cobble bars along shallow, low-gradient reaches of major streams (Ashton et al. 1998). Foothill yellow-legged frogs may breed from early April into early June. A combination of environmental variables such as water temperature, air temperature, and spring runoff (to avoid both late season flooding and recession of waters) likely influence the timing of egg laying (Lind et al. 1992). If flooding occurs, eggs are susceptible to washouts by high flows. If waters recede prematurely, eggs are susceptible to desiccation. The species is known to interrupt egg laying during periods of rain, which is possibly an adaptive response to the threat of late season flooding (Kupferberg 1996). Some authors have observed breeding activity to occur only over a 2 week period within one season (Nussbaum et al. 1983, Leonard et al. 1993). Observations along the Trinity River in California indicate breeding activity can occur over a 3 month period, from April through late June (Ashton et al. 1998).

The most successful egg laying sites are downstream from and within 328 feet of the confluences of lower order tributaries (Kupferberg 1996). Egg laying sites typically are on the lee side of larger rocks, with velocities significantly lower than ambient flow (Kupferberg 1996). The Trinity River study found the majority of egg masses occurred in backwater pools, edgewater pools, and glides adjacent to main channel low gradient riffles and runs (Lind et al. 1992). Egg masses are laid in late spring or early summer, in cobble and pebble substrates, at depths of less than 20 inches, within quiet scour pools or riffles in gentle-gradient streams. Developmental rates vary greatly with water temperature. Eggs in the Trinity river were found to hatch in 27 to 36 days, but may hatch in as little as 5 days (Ashton et al. 1998). Tadpoles attach themselves to the egg mass initially, then to rocks with their specialized enlarged mouths (Corkran and Thoms 1996). Nussbaum et al. (1983) note tadpoles in California metamorphose in 3 to 4 months. Sexual maturity likely occurs in the second spring or summer (Nussbaum et al. 1983). Reproductive success is highly dependent on year-to-year hydrologic variation (reviewed in Ashton et al. 1997).

Diet

Tadpoles forage by scraping algae and diatoms off rock surfaces. They have also been observed feeding on dead tadpoles and dead, open bivalves (Ashton et al. 1998). Adults and juveniles are known to forage on aquatic and terrestrial invertebrates.

Cover requirements

Some shading (>20 percent) seems to be preferred by sub-adults and adults (Ashton et al. 1998). No information is available on canopy closure requirements for wintering habitat.

Dispersal

Little is known about movement and dispersal of the foothill yellow-legged frog (Jennings and Hayes 1994). Observations along the Trinity River indicate adults congregate around breeding pools in late spring and early summer, but later in summer adults were scarce in pools. This may be due to dispersal into vegetation or into tributaries, or just reduced diurnal activity (Ashton et al. 1998). Upstream migration has been observed in recently metamorphosed frogs (Twitty et al. 1967 cited in Ashton et al. 1998), but the overall dispersal range for tadpoles or juveniles is not well understood.

Special habitat needs

It appears rocky, gravelly and sandy river substrates for breeding, and vegetated adjacent terrestrial habitat for dispersal and overwintering is important for the species. See discussion in **Definition of Suitable Habitat** section for more information.

Current legal status

Listing History

The foothill yellow-legged frog is currently identified by the CDFG as a Species of Special Concern throughout the state of California. State endangered status has been recommended in coastal counties south of the Salinas River; State threatened status has been recommended in west slope drainages in the Sierra-Cascade ranges east of the Sacramento and San Joaquin rivers (Jennings and Hayes 1994). It is considered a Federal Species of Concern, but is not currently a Federally listed, proposed or candidate species.

Threats

The primary threats to the foothill yellow-legged frog appear to be altered flow regimes and effects of introduced species. The timing of dam releases or heavy precipitation can wash away egg masses and tadpoles, and disrupt reproduction (Kupferberg 1996, Lind et al. 1996). Other actions such as mining or grazing can also result in significant changes to channel morphology, thereby affecting local populations. All life stages are susceptible to competition and/or predation from introduced fish stocks and bullfrogs (*Rana catesbiana*) (see Ashton et al. 1998 for a review of the literature). Kupferberg (1996) notes the reproductive strategy is clearly linked with predictable winter-floods, summer-droughts, and selection of distinctive channel morphologies: the timing of egg laying appears to be correlated with the end of the seasonal flooding period, and eggs are placed where dessication is less likely to occur.

Conservation needs

Mechanisms which maintain natural flow regimes and do not alter channel morphology should be implemented. Spring water release in dammed river systems should be timed to correspond with high water levels in early spring to avoid the flushing of egg masses and loss of suitable microsite environment selected by adults. Changes in channel morphology can reduce suitable habitats for native species, while increasing suitability for non-natives, typically resulting in increased predation or competition with native species. This has been observed in the main stem of the Trinity River system (Lind et al. 1996). In managed landscapes, the structural diversity of aquatic systems are benefitted by snag and log retention provisions which allow input of these structures into the system. Use of overwintering habitats needs to be investigated.

Status and Distribution

Species

Numbers

Range-wide population data are not known for the foothill yellow-legged frog. However, populations have experienced significant declines, especially in the southern and inland parts of the range (G. Fellars pers. comm., cited in Kupferberg 1996). In southern California, the species has not been observed in or south of the Transverse Ranges since before 1978, and in the southern Sierra Nevada, the species has not been observed for at least 20 years in at least 19 historical localities. Populations appear to be widely scattered along the western slope of the northern Sierra Nevada and the extreme southern Cascades. In the Coast Ranges, north of the Salinas River, significant numbers of this species occur in some coastal drainages, but are still at risk due to various anthropogenic and environmental threats (Ashton et al. 1998).

Distribution

The foothill yellow-legged frog occurs in most Pacific drainages west of the Sierra-Cascade crest from the Santiam River, Marion county, Oregon to Kern county, California. In coastal areas it is found from Douglas county, Oregon to the San Gabriel drainage in Los Angeles county, California. There are records of an isolated population in the Sierra San Pedro Martir, Baja California, Mexico (Leonard et al. 1993). The species range maps for California (Zeiner et al. 1988) overlap portions of fourteen National Forests. The foothill yellow-legged frog is found from sea level in coastal drainages to 6,000 feet in the Sierra Nevada range (Zeiner et al. 1988).

Reproduction

Range-wide reproductive trend data are not available. As discussed above, habitat alterations and increases in predation and competition from introduced species combine to lower reproductive success and recruitment capability for this species.

Suitable habitat

Acreage and distribution

Estimates were made of gross land acreage and perennial stream miles within the portion of the species range addressed by the Pacific Northwest Forest Plan. The total landbase in that area, public and private combined, is 26,802,251 acres, of which 64 percent is in California and 36 percent is in Oregon. The estimate is based on species range maps in Csuti et al. (1997), and Ziener et al. (1988); data on stream mileage were not available. A second estimate was made, within that area, for lands managed by the USFS and BLM in Oregon and California; with estimates based on Table V-G-4 in FEMAT (USDA Forest Service et al. 1993), and on the range maps identified above. Acreage and perennial stream mileage in Table V-G-4 were adjusted by the estimated percentage of each administrative unit (USFS or BLM District) lying within the species' mapped range. The agency totals for both states are 11,478,711 gross acres and 30,916 miles of perennial streams; of which California agency lands account for 43 percent of the acreage and 30 percent of the stream mileage. An unknown portion of this acreage and stream mileage may be unsuitable for the species.

Quality

Habitat quality throughout much of the species range has declined because of artificial changes in river and stream morphologies. Refer to discussion of **Threats** to the species above.

<u>**Northwestern pond turtle**</u>

Species Description

The western pond turtle is a fresh water turtle which occurs in the Pacific States of North America. Two subspecies are recognized: the southwestern pond turtle *(Clemmys marmorata pallida)*, and the northwestern pond turtle (*Clemmys marmorata marmorata*). Western pond turtles are described as a relatively long-lived aquatic habitat generalist occurring in a wide variety of permanent and ephemeral habitats ranging from sea level to over 4,500 feet, with few sightings in upper elevations.

Life history

Definition of suitable habitat

Unless otherwise noted, information in this section was obtained from Holland (1991) or (1994). The subspecies uses upland terrestrial habitat and various aquatic habitats (e.g., rivers, streams, lakes, ponds, vernal pools and other seasonal and permanent wetlands). Favorable habitat is characterized by deep, slow-flowing pools of cool and/or warm water with subsurface cover and emergent basking sites (Holland 1991, 1994; Reese 1996). Reese (1996) found juveniles closely associated with warmer, more lentic waters than adults. These areas may be either permanent or intermittent, with permanent streams supporting larger populations. Along intermittent streams, pools are often used when the stream course dries. In some areas, turtles may be restricted to areas near river banks or in quiet backwaters where the water is slow moving and basking sites and refugia are available. Areas lacking refugia are avoided. Basking sites consist of partially submerged logs, rocks, mats of floating vegetation, or exposed mud banks. In Oregon, the subspecies is found primarily along rivers, large order streams, and wetlands. Sighting records for the coastal Oregon region are approximately equally divided between river, stream, and lake habitats.

The subspecies also uses altered habitats such as farm ponds, canals, mill ponds, and sewage treatment plants. Holland, however, suggests these habitats may not support viable populations of turtles and the populations at these sites may be displaced from other habitats.

Terrestrial habitats are essential for overwintering, estivation, nesting (discussed in section below), and overland movement between aquatic habitats throughout the year. The duration of overwintering, type of habitat used, and the distance of that habitat from water is highly variable. Some individuals may not overwinter, while others may overwinter from October through March or April (Holland and Bury in press). Reese and Welsh (1997) found movement increased in September for upland use, and began again in February, lasting as late as June. Two turtles in Oregon were documented as spending at least 8 months in or near an overwintering site (Holland 1994). Overwintering habitat has been found in undercut areas or holes along watercourse banks, upland grassland, conifer or hardwood habitats, under logs, or duff/litter layers within wooded areas, or in the bottom mud of streams or ponds (Holland 1991, Reese and Welsh 1997). Reese (1996) found percent slope of overwintering locations varied from 0 to 55 percent. Some turtles will not stay in one location during the winter season and will change locations numerous times (Holland 1994, Reese 1996, Reese and Welsh 1997).

Reese and Welsh (1997) found terrestrial movements most commonly occurred during summer and winter along the Trinity River in California. Both sexes, most notably gravid females, exhibited frequent and prolonged use (7 months) of terrestrial habitat. Overwintering locations in this study were all found in upland habitats beyond riparian zones, ranging from approximately 213 to 1,640 feet (mean distance of 666 feet) from the water. Reese (1996), found overwintering distances ranging from 143 to 1,387 feet (mean of 496 feet). Elevations of overwintering sites are variable as well: range from 0 to 521 feet (mean of approximately 140 feet) above the stream. Individuals of the southern subspecies were observed to move over 1,500 feet into upland habitat

(Rathbun et al. 1992, Holland and Bury, in press). Overland movement between watercourses is highly variable and is not well understood (Holland 1994). Some turtles may regularly move distances of over 3 miles within a stream and up to 3 miles overland (Holland 1991, 1994).

Reproduction

The natural history of the northwestern pond turtle varies throughout its range. Sexual maturity in females may vary geographically, but they typically reproduce between 7 to 11 years of age. Clutch sizes varies from 1 to 13 eggs, with clutch size positively correlated with carapace length (Holland 1991, 1994). Females are highly sensitive to disturbance during nesting. Eggs can be laid every year, two clutches in one year, or every other year. Most eggs are laid in June or July (Holland 1994). The time required to excavate a nest site is variable, ranging from 2 hours to multiple efforts over a 24 hour period (Holland 1994). Distance from water to nests is also variable. Of 252 nest sites located in Trinity County, California; Lane, Douglas, Wasco, Counties in Oregon, Holland (1994) found the mean distance of nests to water was approximately 160 feet (range = < 10 feet to 1,318 feet). The majority of these sites were less than 230 feet from the water. Mean distance from water in broad river valleys in this study were found to be further than in narrower river canyons. For example, mean nest to water distance in the Willamette drainage was 282 feet, compared with 94 feet in the South Umpqua. Along the Trinity River, Reese (1996) found nest sites and potential nest sites were located at a mean of 253 feet (range 6.5 to 476 feet) from the water's edge. Reese and Welsh (1997) documented one nest which was located approximately 100 feet from the water's edge. Nest sites are typically on south or south west facing slopes dominated by grasses or herbaceous annuals in dry, well-drained soils with significant clay/silt content and low slopes between 0 and 60 percent, with most nests on slopes 25 percent. Hatchlings require shallow water habitat characterized by relatively dense submergent or short emergent vegetation for foraging and basking (Holland 1985, Jennings and Hayes 1994). Reese (1996) reported higher proportion of juveniles/adults in ponds rather than riverine habitats.

Diet

The diet of the species is variable, and dominant items in the diet may vary from area to area. Literature was summarized in Holland (1994) and Holland and Bury (in press). The species consume primarily aquatic insects and crustaceans, fish, and small amphibians. These items are taken as carrion or prey. Pond turtles also eat aquatic vegetation, but use of plants appears to be relatively infrequent.

Cover requirements

Pond turtles favor aquatic habitats with an abundance of basking sites, underwater refugia, and vegetative canopy. A high correlation between turtle abundance and number of basking sites such as logs and boulders was observed along a northern California stream (Bury 1972, Holland and Bury in press). Aquatic vegetation such as tule, cattails, or algae are often used for resting or hiding (refer to **Definition of suitable habitat**). Reese (1996) found that turtle locations are associated with vegetated banks with greater frequency than expected from their availability and that the majority of overwintering sites exceeded 50 percent canopy closure.

Dispersal

Dispersal of individuals is primarily restricted to aquatic habitats. Post-emergence terrestrial movements of hatchlings are not well known, however hatchlings are known to overwinter in the nest (Holland 1994, Reese and Welsh 1997). Juveniles sampled on the Trinity River traveled a mean weekly distance of 65 feet.

The amount and distance of movement by adults is highly varied. Environmental stress, disturbance, or local conditions such as the amount and type of available habitat may influence the movement of adults (Holland 1994). Reese and Welsh (1997) found the size of traditionally protected buffer zones along rivers did not sufficiently incorporate areas used by dispersing turtles. Turtles use terrestrial areas ranging from 213 to 1,640 feet from water.

Special habitat needs

The presence of suitable nesting and overwintering habitat, vegetatively linked wetlands, basking sites, and underwater refugia appear to be important for the northwestern pond turtle. Refer to **Definition of suitable habitat** and **Reproduction** sections for more information.

Current legal status

Listing history

The northwestern pond turtle is a Federal species of concern, USFS sensitive species, and State species of special concern and protected species (CDFG 1998a, USDA Forest Service 1998). It is not currently Federally recognized as a candidate, proposed, or listed species.

Threats

Populations of the subspecies are currently threatened by the following factors: habitat destruction and fragmentation due to agriculture, timber management, livestock grazing practices, damming, and water diversions, and other human-related disturbance. These factors vary in magnitude within different portions of the subspecies' range (Holland 1994, Jennings and Hayes 1994, Reese 1996, Reese and Welsh 1998). Other human related factors such as collisions with vehicles, hunting, shooting, and water contamination have resulted in mortalities throughout the range. Introduced predatory and/or competitor species including bullfrogs, sunfish, and bass can threaten young turtles in the aquatic environment. Terrestrial predation by racoons, skunks, and coyotes has been observed to significantly affect nest sites (Holland 1994).

The relatively long period of time (about 7 to 11 years in California) required to reach reproductive maturity makes the population's growth rate particularly vulnerable to changes in juvenile and adult survival (Jennings and Hayes 1994). A range-wide concern for northwestern pond turtles involves juvenile recruitment, as many populations are composed primarily of aging adults (Holland 1991, 1994, Reese and Welsh 1997). This situation is most severe in the upper Klamath River, Oregon (D. Holland, pers. comm., October 21, 1998). While populations may appear stable, in the long-term, they may not be. Without adequate recruitment into the population, adults that are lost from the population may not be replaced.

Conservation needs

Holland and Bury (in press) summarize the conservation needs of the northwestern pond turtle. Protection of existing populations and suitable habitat, and continuation or expansion of current ecological monitoring programs are the primary conservation needs. Wetlands should be linked to facilitate dispersal and limit population fragmentation (Burke and Gibbons 1995, cited in Reese 1996). Implementing management strategies which address the functioning of entire watersheds have a higher likelihood of providing adequate protection for the northwestern pond turtle (Reese and Welsh 1997).

Mitigation should not rely on translocation and captive breeding programs for a number of reasons, discussed in detail in Holland (1994). Translocated turtles have been known to return to original sites, thereby subjecting them to increased stress and risk of mortality. The effectiveness of these efforts has been questioned (Dodd and Seigel 1991). The translocations of genetically different individuals into a different population may result in an elimination or dilution of gene pools adapted to specific conditions (Holland 1994); or these efforts can lead to disease transmission, either to the resident or translocated population.

The Northwest Forest Plan clearly will contribute to improvements in riparian systems and it is expected benefits to the turtle will be achieved where they exist on Federal land. However, because many turtles are found in low-lying areas, much of their range occurs on private lands not subject to the provisions of the Northwest Forest Plan; therefore, long-term protection measures on private lands will play a significant role in the conservation of this subspecies.

Status and distribution

Species

Numbers

The western pond turtle is declining in numbers throughout its range, with significant declines noted in the northern and southernmost portions (Holland 1991, 1994, Holland and Bury in press). Although the turtle was formerly widespread and abundant in many aquatic habitats on the west coast, it is now common in only a small fraction of its original range (Holland and Bury, in press). Many populations throughout its range contain a significantly higher proportion of adults to juveniles (Holland 1991, 1994; Reese 1996). Population levels and trends are difficult to determine with certainty for the following reasons: habitat use varies geographically; distribution within habitats varies within or between seasons; many populations are adult-dominated; and historical baseline populations for some areas are not available.

Population estimates for the northwestern subspecies are not available, but some estimates of population size are available for certain areas. For example, Holland (1991, 1994) estimates the total population in the state of Washington is only as high as 110 individuals, and the Willamette River drainage in Oregon contains between 1,500 and 2,400 individuals. Holland indicates the Willamette population represents a decline of 96 to 98 percent from levels conservatively estimated to be present in the late 1800's. In northern California, Reese (1996) estimates a population of 1,318 pond turtles in all 16 study reaches in the Trinity River system.

Distribution
The northwestern pond turtle ranges west of the Sierra/Cascade Crest, from approximately the American River in California, north to the vicinity of Puget Sound (Stebbins 1985). In some areas, disjunct and isolated populations are known to occur (Holland 1991). The taxon also includes three distinct evolutionary groups, based on morphological variation (Holland 1992, cited in Holland and Bury in press): Columbia River form; Puget Sound-Willamette River to central California form; and the central coast of California to Baja California form.

Most northwestern pond turtles occur in major river drainages such as the Klamath, Rogue, Umqua, Willamette, and Columbia River systems, but is uncommon or absent along the north and central coast of Oregon, and the north coast of California (Holland 1991, 1994). Dense, local concentrations may occur in slow moving streams, ponds, or shallow lakes, but in most areas the turtle is uncommon or occurs in clusters of individuals along segments of streams (Holland and Bury, in press).

Specific to northern California, Holland (1991) noted that northwestern pond turtles are widely distributed at low densities, primarily within lowland areas of the Klamath River and tributaries such as the Eel, Russian, Gualala, and Mad Rivers. Within this system, they are found in watercourses of all sizes, but are most abundant in small to medium-sized, shallow, warm streams. These tributaries contain up to a few hundred individuals with clusters of turtles more likely to occur inland from the coast. The Hayfork system may have several thousand individuals, a uniquely high population level (Holland, pers. comm., October 21, 1998). Along the northern Pacific coast, the northwestern pond turtle occurs in relatively low numbers in lower reaches of river drainages. Densities tend to increase inland along upper reaches which contain more suitable habitat elements (B. Bury, pers. comm., August 4, 1998, D. Holland, pers. comm., October 21, 1998).

Suitable habitat
Acreage, distribution, and quality
Data on the range-wide quantity, quality, or distribution of habitat are not available for the purpose of this consultation. Suitable aquatic habitat today, however, is currently highly fragmented and disjunct, compared to historic conditions in the subspecies' range. In some areas, Holland (1991) notes large areas of apparently suitable habitat that are unoccupied, which indicates that the increasing distances between suitable habitat areas are becoming an impediment to dispersal. In FEMAT (USDA Forest Service et al. 1993) table V-3 shows the 50-year declines in the frequency of large, deep pools (>50 square yards and > 6 feet deep) in selected river basins with initial surveys in 1935-40, and recent surveys in 1987-92. From that table we selected the seven rivers, with a total sample of 110.8 survey miles, in coastal Washington and Oregon for separate analysis. This subset was most similar to the action area in terms of climate, coastal location and land use history. Reductions in pool frequency (pools per stream mile) among the seven rivers diminished from zero to 94 percent in the 50-year period, with a weighted mean pool loss of 74.0 percent. Two of the seven rivers had less than a 50 percent reduction; and they accounted for only 14.7 percent of the total survey mileage.

Tailed frog

Species description

The tailed frog is endemic to cold, fast-flowing water in perennial, mostly non-fish-bearing, streams in the Pacific northwest (Welsh et al. 1993), and is the only North American frog highly specialized for life in cold, clear mountain streams (Nussbaum et al. 1983). Larvae range in size from 0.2 to 1.0 inches snout-vent length; adults range from 0.9 to 2.0 inches snout-vent length (Corkran and Thoms 1996). The tail-like appendages on males are the genitalia (Nussbaum et al. 1983). Larvae are fully aquatic. Adults are strongly aquatic, but are known to forage upland up to 100 yards from water during rainy periods (Noble and Putnam 1931).

Life history

(1) Reproductive Potential. The species has the longest larval period (2 to 4 years) and longest timespan to sexual maturity (7 years from hatching to breeding) of any North American frog species. Lifespan is estimated at 15 to 20 years (Daugherty and Shelton 1982a). However, Wallace and Diller (1996) have observed larvae that metamorphosed in 1 year in northwestern California. In coastal areas, females lay eggs annually; in inland areas they lay eggs every other year (Nussbaum et al. 1983). Metter (1964) reported clutch sizes of 37 eggs per female in coastal areas and 68 eggs per female in inland areas.

(2) Habitat Specificity. The tailed frog is associated with highly specialized habitat features. Welsh et al. (1993) identified eleven habitat attributes at three hierarchical scales: (a) landscape; (b) macrohabitat (adjacent land vegetation attributes), and (c) microhabitat (aquatic attributes). Landscape variables included (a1) latitudinal, (a2) longitudinal, and (a3) elevational limits that define the species range within the study area. Macrohabitat variables associated with frog presence are as follows (with mean, minimum and maximum observed levels for occupied habitat patches): (b1) mature to old-growth structure, with a mean stand age of 335 years (range 12 to 941 years); (b2) presence of large trees greater than 21 inches dbh with a mean of 34 trees-per-acre (range of 0 to 97 trees-per-acre); (b3) mean tree canopy closure is 86 percent (range 50 to 97 percent); and (b4) ground-level cover is a combination of low grass cover plus high cover percent of mosses, herbs, ferns and rock. Sites with suitable macrohabitat are very limited in size and isolated from other patches of suitable habitat.

Aquatic microhabitat variables associated with frog presence are as follows (with mean, minimum and maximum observed levels for occupied habitat patches): (c1) mixture of scour pools (mean 14 percent, range 0 to 44 percent cover), stream edge (mean 2.5 percent, range 0 to 12 percent) and stream run, or thalweg (mean 9.6 percent, range 0 to 34 percent) in perennial, cold water streams; (c2) coarse substrates are a mix of boulders (mean 15 percent, range 0 to 76 percent cover), gravel (mean 11 percent, range 2 to 24 percent) and decayed vegetation (mean 5 percent, range 0 to 20 percent); (c3) fine substrates are non-filamentous algae (mean 49 percent, range 2 to 88 percent), and moderate- to low- sediment embeddedness (mean 35 percent, range 5 to 95 percent); (c4) fast-moving streams (mean discharge 72.45 cubic feet per second, range 1.1 to 410.0 cubic feet per second), with mean channel width of 10.5 feet and a range of 2.6 to 36.3 feet. (c5) Water temperature was not measured by Welsh et al. (1993); the b2 and b3 variables above were used as surrogates, which correlate directly to riparian zone soil temperature and indirectly to adjacent water temperature. The species is known to have the narrowest thermal tolerances and lowest maximum temperature limits of any North American frog (Davidson 1993). Published thermal tolerances, measured in the field, by life stage are: eggs (41 to 65°F), larvae (up to 66°F), and adults (32 to 61°F), with incipient lethal temperatures for adults in the range of 74 to 75°F (reviewed in Davidson 1993). Several interactions among the presence variables, above, are also statistical predictors of abundance on occupied sites. Taken together, all the presence variables indicate a species for which sites with suitable macrohabitat are very limited in size and isolated from other patches of suitable habitat. The authors did not provide statistics for the mean area of suitable sites they evaluated, but they are mostly on the order of several square yards to several tens of square yards each (L. Ollivier, pers. comm., 1998).

(3) Dispersal and Site Fidelity. No information was available on larval dispersal. Adults are highly sedentary. Year-to-year mark-recapture data for adults show a statistically significant decrease in movement with increasing age. Annual movement rates were (males) 102 feet per year at age 5, declining to 31 feet at age 9; and (females) 154 feet per year at age 5, declining to 26 feet at age 11 (Daugherty and Shelton 1982b). Most of the movement (both sexes) was upstream. This movement pattern is consistent with a scenario of restricted gene flow and a highly fragmented population (Daugherty and Shelton 1982b).

(4) Population Structure. Tailed frogs live in highly subdivided populations and are considered a metapopulation (Welsh et al. 1998). Refer to the discussions above on dispersal, and below on genetic variability.

(5) Genetic Variability. Metter (1967) and Metter and Pauken (1969) report that tailed frogs exhibit a progressive reduction in gene flow between subpopulations that has probably been occurring throughout the last 10,000 years. They attribute this to climate warming since the last ice age, followed by a shrinking habitat base and progressive disjunction of subpopulations throughout the species range. Isolated sub-populations are highly susceptible to population decline and local extirpation due to "founder effects" and diminished intra-population genetic diversity. They are even more susceptible, at the genetic and population levels, to human-caused reductions in habitat. For a discussion of the implications of genetic variability in animal populations, see Allendorf and Leary (1986). For a discussion of how small populations and low gene variability interact in extinction processes see Gilpin and Soule (1986).

(6) Food Web Position. Nussbaum et al. (1983) identify Pacific giant salamanders (*Dicamptodon tenebrosus*) as common predators of tailed frogs. Aquatic microhabitat associations for the two species overlap (Welsh et al. 1993). Some of the habitat associations for tailed frog (e.g., sediment embeddedness) are considered a compromise between preferred attributes and predator avoidance strategies (Welsh et al. 1993). Dispersal capability of the tailed frog is probably constrained by predation.

Current legal status

Listing history

The species is currently identified as a species of special concern by the CDFG.

Current Known Range

Within the tailed frog's range in British Columbia, Washington and Oregon, it occurs in forested areas between the Cascade crest and the Pacific coast, as well as a disjunct population in the Blue Mountains. In Idaho it occurs in forested areas north of the Snake River. In Montana it occurs in forested areas west of the continental divide. In California, it occurs in forested areas in Del Norte, Humboldt, Siskiyou, Trinity, Shasta, Tehama, and Mendocino Counties (Zeiner et al. 1988, Nussbaum et al. 1983).

Threats

The primary threat to the tailed frog is the loss of suitable habitat, resulting in further fragmentation of an already subdivided population. Daugherty and Shelton (1982b) have reported that movement patterns indicate a fragmented population. Metter (1967) and Metter and Pauken (1969) have reported that the species is also genetically fragmented throughout its range. These attributes suggest that the species would have limited success in colonizing "unfilled" suitable habitats in an intensively managed forest environment. A comparison of presence/absence data between forest age classes by Welsh et al. (1993) indicates that this recolonization is limited. Progressive losses of habitat patches exacerbate the problem. With fewer suitable habitat patches available, average distance between patches becomes greater and eventually exceeds the dispersal capability of the species. Over time, disproportionately fewer patches are occupied.

The suitability of a habitat patch is a function of numerous physical attributes (outlined in the **Life history** section above), some with relatively narrow tolerances so degradation of any single attribute can cause a loss of the patch. Below are some proposed mechanisms for habitat loss.

(1) Loss of stream channel features: This species is highly dependent on coarse streambed substrates (e.g., rock, cobble, and especially boulders and gravel). In functional channel morphological processes, these substrates are collected largely by instream blockages caused by downed trees. Over time these coarse substrate features are lost due to, for example, decomposition and abrasion of the downed wood, or sedimentation, or flood scouring. Recruitment levels of instream coarse woody debris (CWD) must be sufficient to replace these substrate features at a rate greater than, or equal to, the rate at which they are lost (Sedell et al. 1988, Welsh et al. 1998). CWD depletion in upland riparian areas can interrupt this feature-forming process and cause long term declines in suitable habitats downstream.

(2) Obliteration by debris flows: Habitat patches are obliterated directly and entirely by debris flows from mass failures upslope (L. Ollivier, pers. comm., 1998).

(3) Sediment infusion: Habitat utility can be lost, almost immediately, by fine sediment infusions. Welsh and Ollivier (1998) found marked (and highly significant) reductions in tailed frog larvae in streams recently impacted with sediments. They concluded that the suspended sediment scoured

freshwater diatoms from the rocky substrate and eliminated the primary food source for grazing tailed frog larvae. Sediments fill interstitial spaces in the coarse substrates and obliterate rearing habitat for larvae (Welsh et al. 1998), even though the high-velocity stream features they use (step-runs and riffles) are more resistant to sediment accumulation (Welsh and Ollivier 1998). Intensive timber harvesting combined with inadequate riparian protection upstream will threaten any downstream habitat patches.

(4) Degradation of microclimate: Favorable conditions of air temperature and humidity in streamside areas can be disrupted by losses of adjacent forest canopy cover. Tailed frogs have a low tolerance (Welsh et al. 1993) for canopy closures of less than around 80 percent (as measured by spherical densiometer).

(5) Degradation of water temperature: The tailed frog has the narrowest thermal tolerances and the lowest maximum temperature levels of any North American frog species (reviewed in Davidson 1993). Warm water infusions that may result from harvesting, or other disturbances, combined with inadequate riparian protection in upstream reaches may negate the suitability of a habitat patch that is suitable in every other respect.

<u>Conservation Needs</u>
(1) Long-term maintenance of favorable recruitment levels of instream CWD. This is needed to maintain desired channel morphological processes in which new coarse substrate features are created, over time, to replace those lost to sedimentation and/or the decomposition and weathering of existing reservoirs of instream CWD (see Threats, item 1).

(2) Mass-wasting prevention measures, including limitations on harvest around headwalls, inner gorges and unstable areas, and where mass wasting hazard is high to extreme (see Threats, item 2).

(3) The sediment storage capability of all streams (perennial, intermittent, ephemeral, fish-bearing and non-fish-bearing) must be maintained at favorable levels in perpetuity by maintaining adequate CWD recruitment sources in riparian areas. Road management (construction, use limits, repair, and upgrades) should be geared to minimize sediment runoff (see Threats, item 3).

(4) Maintenance of favorable levels of canopy closure in riparian management zones (see Threats, item 4).

(5) Maintenance of favorable water temperatures, not only in stream reaches where suitable habitats are found (mostly Class II reaches), but upstream as well in all potential warm water source areas (see Threats, item 5).

Status and distribution
Species
Numbers

No range-wide census data are available for this species.

Distribution
Populations are known to be highly subdivided due to specialized habitat associations and other population and community-level factors (see Metter 1967, Metter and Pauken 1969, Daugherty and Shelton 1982b, Corn and Bury 1989, Welsh et al. 1993, Welsh et al. 1998). There is strong converging evidence among various studies that, in the absence of significant mitigation measures, intensive land management practices are exacerbating the fragmentation of populations.

Reproduction
No range-wide reproductive data are available for this species, but see the discussion on reproductive potential in the **Life history** section, above.

Suitable Habitat
Suitable stream habitats are found in montane forestlands of western and north-central Oregon, western and south-central Washington, western British Columbia, northern Idaho, western Montana and northwestern California (Stebbins 1985, Corkran and Thoms 1996), comprising approximately 33.46 million acres (52,286 square miles). Under the FEMAT (USDA Forest Service et al. 1993) analysis, approximately 56 percent (18.74 million acres, or 29,280 square miles) of the species range is on Federal land. Twenty-four percent of the Federal land (4.5 million acres, or 7,027 square miles) is matrix; seven percent (1.31 million acres, or 2,050 square miles) is in Adaptive Management Areas; the remainder (23.09 million acres, or 36,077 square miles) is in reserved or withdrawn status.

Acreage, Distribution and Quality of Habitat
No range-wide data are available on quantities, distribution and quality of suitable habitat patches. However, FEMAT (USDA Forest Service et al. 1993) notes that ranges of numerous aquatic and forest floor amphibian species in the Pacific northwest have been reduced.

<u>Southern torrent salamander</u>
Species description Four separate torrent salamander species (including the southern torrent salamander) are currently recognized (Good and Wake 1992) that were formerly classified as Olympic salamander (*Rhyacotriton olympicus*). All four torrent salamander species are characterized by their small adult size (1.2 to 2.2 inches snout-vent length) (Corkran and Thoms 1996), extended juvenile life stage, and highly specific habitat associations (Welsh and Lind 1996). Larvae are fully aquatic, adults are semi-aquatic and both are highly sedentary.

Life history
Several population-level and community-level factors probably contribute to this species' apparent vulnerability. These include reproductive potential (see 1, below), site fidelity (see 3, below), specialization and insularization of habitats (see 2 and 4, below), spatial and genetic structure of populations (see 4 and 5 below), and food web niche (see 6, below). Major issues of concern at the population and community levels are: (a) the apparent vulnerability of this species and its

habitats to intensified forest management regimes; and (b) the limited ability of this species to exploit available and suitable habitats in an intensively managed forest landscape.

(1) Reproductive Potential. Embryos (eggs) require approximately one year to hatch. Aquatic larvae require 3.0 to 3.5 years to metamorphose. Metamorphosed subadults require an additional 1.0 to 1.5 years to reach sexual maturity (Nussbaum and Tait 1977). The total time span from conception to sexual maturity is 5.0 to 6.0 years. Total lifespan is not known. Behler and King (1979) and Good and Wake (1992) report clutch sizes of 2 to15 eggs. However, Welsh and Lind (1992) found clutch sizes of only 2 and 6 eggs in the two gravid females they captured in a population study in northwestern California. Juvenile-to-adult ratios ranged from 4 to 1 in the spring to 7 to1 in the fall (Welsh and Lind 1992), though it should be recognized that the juvenile cohorts, at any given time, represent three years of reproductive effort. Suitable habitats for the southern torrent salamander are frequently unoccupied. Occupancy rates on suitable habitats can range from almost 80 percent in relatively pristine parklands to less than 20 percent on intensively managed timberlands (Welsh et al. 1998).

(2) Habitat Specificity. Welsh and Lind (1996) developed a habitat association model for this species and identified thirteen variables at three hierarchical scales: (a) landscape, (b) macrohabitat (adjacent land vegetation attributes), and (c) microhabitat (aquatic attributes). Landscape variables included (a1) latitudinal, (a2) longitudinal and (a3) elevational limits that define the species range within the study area. Macrohabitat variables associated with salamander presence, and their threshold (minimum) levels, are as follows (ranges, when given, are at the 95 percent confidence level): (b1) vegetation series is Douglas-fir or redwood dominated forest; (b2) seral stage is mature to old-growth structure, except coastal areas; (b3) large trees, range from 9 to 15 trees per acre, or more, greater than 21 inches dbh; (b4) total tree canopy closure is 83 to 95 percent; and (b5) ground-level cover is a combination of low numbers of stumps plus low cover percent of grasses plus high cover percent of mosses. Because of the number of correlated habitat features, occupied habitat patches are very limited in size and isolated from other patches of suitable habitat, ranging from a few square yards in extent to several tens of square yards (L. Ollivier, pers. comm., 1998). Microhabitat variables are as follows (ranges are also at the 95 percent confidence level, except water temperature): (c1) aquatic types are cold-water springs and seeps along first-through-third order streams (roughly equivalent to middle and upper reaches of Class II streams); (c2) coarse substrates are cobble mixed with pebble and gravel (25 to 36 percent surface coverage); (c3) fine substrates are sand and other fine particles mixed with organic material (18 to 33 percent surface coverage); (c4) aquatic condition is shallow, slow-flowing stream or seepage with cold, clear water; and (c5) water temperature is in the range of 15.0 to 43.7 °F. Taken together, all of the thirteen variables were found to be reliable predictors of presence, and two microhabitat variables (coarse substrates and fine substrates) were found to be correlated with abundance on occupied sites.

(3) Dispersal and Site Fidelity. The species is highly sedentary, but larvae are significantly more mobile than adults. Mark-recapture data show that larvae move an average of 0.6 to 7.0 feet per year between captures; and adults move 0.3 to 3.3 feet per year. The directions of larval

movement are downstream (52 percent of all observations), upstream (32 percent) and stationary (16 percent); while the directions of adult movement are upstream (50 percent), downstream (38 percent) and stationary (13 percent) (Welsh and Lind 1992). The data suggest that larval dispersal is the most likely means of gene flow between local populations. However, since extant populations are highly subdivided and insularized, it is probable that significant dispersal events are episodic and only occur during the rainy season, when habitats are more interconnected by high water (Welsh and Lind 1992).

(4) Population Structure. Southern torrent salamanders live in highly subdivided populations and are considered a metapopulation. Welsh and Lind (1992) examined the frequency of suitable habitats, and the frequency of occupied habitats, throughout the species range in northwestern California. They located suitable microhabitats at a rate of 0.18 microhabitat sites per stream mile and occupied suitable microhabitats at a rate of 0.11 occupied habitat areas per stream mile.

(5) Genetic Variability. Good and Wake (1992) report that torrent salamanders exhibit some of the highest inter-population genetic diversity ever observed between proximate populations of a vertebrate species. Taken with the population structure studies in (4) above, Welsh et al. (1998) suggest that fragmentation and isolation have long been part of their evolutionary history. However, small, isolated sub-populations are highly susceptible to population decline and local extirpation due to "founder effects" and diminished intra-population genetic diversity. For a discussion of the implications of genetic variability in animal populations, see Allendorf and Leary (1986). For a discussion of how small populations and low gene variability interact in extinction processes, see Gilpin and Soule (1986).

(6) Food Web Position. Nussbaum et al. (1983) identify Pacific giant salamanders as common predators of southern torrent salamanders. Both species are proximate and have an affinity for cold, highly oxygenated water, but the giant salamander is much more a habitat generalist and readily uses forest floor habitat. However, little overlap exists in aquatic microhabitats between the two species. The giant salamander is mostly associated with intermixed cobble, gravel and woody debris substrates (using the latter for stalking cover), where the torrent salamander is found in intermixed cobble, gravel and sand substrates with no woody debris (Welsh and Ollivier 1998). The authors suggest that this habitat partitioning may be part of a predator-avoidance strategy by torrent salamanders. If true, this places severe constraints on the ability of southern torrent salamanders to emigrate out of degraded habitat patches in search of unoccupied suitable habitats.

Current legal status

Listing history

The species is currently identified by CDFG as a species of special concern. Listing petitions have been submitted to the State and the FWS. The CDFG has recommended against State listing (Brode 1995). The FWS, in their 90-day petition finding, concluded there was substantial evidence indicating that listing may be warranted. The agency initiated a 12-month status review which was delayed by the listing moratorium, and is now in progress (USDI Fish and Wildlife Service 1995d).

Current Known Range

The current known range is as follows: in Oregon is from the southern fringe of Tillamook County (northern extent of range), southward through Lincoln, Benton, Lane, Douglas, Coos and Curry Counties (Corkran and Thoms 1996, Nussbaum et al. 1983, Stebbins 1985); and in California through all of Del Norte and Humboldt counties, the western fringes of Siskiyou and Trinity Counties, and the northwestern two-fifths of Mendocino County (southern extent of range) (Zeiner et al. 1988, Brode et al. 1997).

Threats

The primary threat to the species is the loss of suitable habitat localities, resulting in further fragmentation of an already subdivided population. Suitable habitat patches are small (on the order of a few square yards each), and widely dispersed (0.18 patches per stream mile in one estimate by Welsh and Lind 1992). Interactions between subpopulations are apparently very weak as indicated by movement data (3.3 to 7.3 feet per year for adults and larvae, respectively) from Welsh and Lind (1992) and by gene distributions that indicate a long history of isolation of subpopulations throughout the species range. Good and Wake (1992) report low variability within subpopulations combined with high variability between subpopulations. These population-level attributes suggest that the species would have limited success in colonizing "unfilled" suitable habitats, which is corroborated by the field observations of Welsh and Lind, 1992. Progressive losses of habitat patches exacerbate the problem through the following proposed mechanisms: fewer suitable habitat patches are available, average distance between patches becomes greater and exceeds the dispersal capability of the salamander, and, over time, disproportionately fewer patches are occupied (Welsh et al. 1998).

The southern torrent salamander has a highly subdivided population, low dispersal capability, and (apparently) a relatively low reproductive potential. One generalization about this set of attributes is that the species is highly susceptible to further fragmentation of its habitat and population. As habitat patches are lost and remaining patches become isolated, it becomes increasingly difficult for sedentary species to recolonize unoccupied patches of suitable habitat (see Wiens, 1996; Hanski, 1997).

Welsh et al. (1998) provide a test of this prediction by comparing four "presence/absence" data sets on suitable habitats in northern California. The four databases were: (a) a systematic sample of public and private forestlands in Humboldt, Trinity and Del Norte Counties used in Welsh and Lind (1996); (b) reserves and parklands in Mendocino County (Welsh, 1990); and (c) & (d) private industrial timberlands in the Mattole River watershed of Humboldt County (Welsh and Hodgson, unpublished data, reviewed in Welsh et al. 1998) and throughout Mendocino County (unpublished reports from landowners). The results are shown as the percentage of suitable sites that are occupied, and are as follows: Systematic Sample – 62.3 percent (n = 53); Mendocino Reserves – 76.9 percent (n = 13); Mattole Industry Land – 29.0 percent (n = 31); and Mendocino

Industry Land – 18.9 percent (n = 53). The results are consistent with the test prediction; reserved lands have the highest occupancy rates, intensively managed lands have the lowest, and the systematic sample of public and private lands have an intermediate rate greater than the mean. Q-test results showed that all differences were statistically significant (Welsh et al. 1998).

The suitability of a habitat patch is a function of numerous physical attributes (see the **Life history** section for this species), some with relatively narrow tolerances so degradation of any single attribute can cause a loss of the patch. Five mechanisms for habitat loss are proposed and are discussed more fully under the corresponding section on the tailed frog. They are: (1) loss of stream channel features; (2) obliteration by debris flows; (3) sediment infusion; (4) degradation of microclimate; and (5) degradation of water temperature.

<u>Conservation Needs</u>

Five conservation needs, related to the five mechanisms for habitat loss are listed below (also see the corresponding discussion on the tailed frog). They are: (1) long-term maintenance of favorable recruitment levels of instream CWD; (2) prevention or minimization of mass wasting events; (3) maintenance of sediment storage capability of all streams; (4) maintenance of favorable levels of canopy closure in riparian management zones; and (5) maintenance of favorable water temperatures, in suitable habitat areas and in potential warm water sources upstream.

Status and distribution

Species

Numbers

No range-wide census data are available for this species.

Distribution

Populations are known to be highly subdivided due to specialized habitat associations and other population and community-level factors, but this inference is based on localized studies (e.g., Corn and Bury 1989, Welsh 1990, Welsh and Lind 1992). There is strong converging evidence among various studies that, in the absence of significant mitigation measures, intensive land management practices are exacerbating the fragmentation of populations.

Another hypothesis for low rates of habitat occupancy (by the southern torrent salamander) has been proposed by Diller and Wallace (1996); that the species is associated with steep-gradient streams. However, the same stream survey at Prairie Creek Redwoods State Park, cited by Welsh, et al. (1998) and described in the tailed frog discussion, also included experimental controls for stream gradient, and revealed no correlation with salamander abundance or presence. Again, the FWS gives greater weight to the alternative explanations (Welsh, et al., 1998) that steep gradients and consolidated parent geology are more resistant to perturbation, so if the species is found predominantly on these sites, then it is an indication that suitable habitat is being lost on the more vulnerable low-gradient and unstable stream reaches. Further support for this interpretation was found in Corn and Bury (1989) who studied southern torrent salamander distribution in harvested and unharvested stream reaches in the Oregon Coast Range. They found that salamander presence was restricted to steep gradients only on the harvested reaches, but found no correlation of gradient and presence on the unharvested reaches.

Reproduction
No range-wide reproductive data are available for this species, but see the discussion on reproductive potential in the **Life history** section, above.

Suitable Habitat
Suitable stream habitats are found in montane forestlands of western Oregon and northwestern California, comprising approximately 10.45 million acres (16,325 square miles). According to the FEMAT (USDA Forest Service et al. 1993) analysis, approximately 37 percent (3.87 million acres, or 6,040 square miles) of the species range is on Federal land. Twenty-five percent of the Federal land (0.97 million acres, or 1,510 square miles) is matrix; six percent (0.23 million acres, or 362 square miles) is in Adaptive Management Areas; and the remainder (2.67 million acres, or 4,168 square miles) is in reserved or withdrawn status.

Acreage, Distribution and Quality of Habitat
No range-wide data are available on quantities, distribution and quality of suitable habitat patches. However, FEMAT (USDA Forest Service et al. 1993) notes that ranges of numerous aquatic and forest floor amphibian species in the Pacific northwest have been reduced.

Northern California ESU steelhead

Species description

The steelhead is a salmonid native to the Pacific drainages of North America and Asia. It is the anadromous form of the rainbow trout (also called "redband" or "golden"), but its ocean-going behavior differentiates steelhead from the resident form of *O. mykiss*. Steelhead also attain a larger size (more than 30 pounds) than rainbow trout. In coastal populations, it is unusual for the two forms (steelhead and rainbow) to coexist; they are usually separated by a migration barrier. In inland populations, co-occurrence of the two forms appears to be more common (Busby et al. 1996). Steelhead are heavily spotted with irregularly shaped spots both above and below the lateral line (Behnke 1992). They are presently distributed from the mouth of Malibu Creek, California, north and west along the Pacific coast to the Kamchatka Peninsula. In some years, steelhead may be found as far south as the Santa Margarita River, in San Diego County (McEwan and Jackson 1996). Historically, steelhead were distributed throughout the North Pacific Ocean from the Kamchatka Peninsula in Asia to the northern Baja Peninsula in North America. However, during this century, over 23 indigenous, naturally reproducing stocks of steelhead are believed to have been extirpated, and many more are thought to be in decline in coastal and inland streams in Washington, Oregon, Idaho, and California. The steelhead status review (Busby et al. 1996) identified 15 distinct ESUs in these four states. The action area is within the Northern California ESU.

Life History

General life history information for steelhead is summarized below. Further detailed information is available in the status review of West Coast steelhead from Washington, Idaho Oregon, and California (Busby et al. 1996), the NMFS proposed rule for listing steelhead (61 FR 41541), the NMFS status review for Klamath Mountains Province steelhead (Busby et al. 1994), the NMFS final rule listing the Southern California Coast, South Central California Coast, and the Central California Coast ESUs (62 FR 43937), and the NMFS final rule listing the Lower Columbia River and Central Valley ESUs (63 FR 13347).

Biologically, steelhead can be divided into two reproductive ecotypes, based on their state of sexual maturity at the time of fresh water entry and the duration of their spawning migration. Stream maturing, commonly called summer steelhead, enter fresh water in a sexually immature condition and require several months to mature before spawning. Ocean maturing, or winter steelhead, enter fresh water in a sexually mature state and spawn soon after river entry. The most widespread run type of steelhead is the winter (ocean maturing) steelhead, while summer (stream maturing) steelhead (including spring and fall steelhead in southern Oregon and northern California) are less common. South of Cape Blanco, Oregon, summer steelhead are known to occur in the Rogue, Smith, Klamath, Trinity, Mad, and Eel Rivers, and in Redwood Creek (Busby et al. 1996).

Summer steelhead enter fresh water between May and October in the Pacific Northwest (Busby et al. 1996). They require cool, deep holding pools during summer and fall, prior to spawning (Nickelson et al. 1992). Summer steelhead migrate inland toward spawning areas, overwinter in the larger rivers, they resume migration in early spring to natal streams where they spawn (Meehan and Bjornn 1991; Nickelson et al. 1992). In contrast, winter steelhead enter freshwater between November and April in the Pacific Northwest (Busby et al. 1996), migrate to spawning areas, and spawn in late winter or spring (Nickelson et al. 1992). Some winter steelhead adults do not enter coastal streams until spring, just before spawning (Meehan and Bjornn 1991).

There is a high degree of overlap in spawn timing between populations, regardless of run-type (Busby et al. 1996). California steelhead generally spawn earlier than steelhead in northern areas. Both summer and winter steelhead in California generally begin spawning in December, whereas most populations in Washington begin spawning in February or March. Among inland steelhead populations, Columbia River populations from tributaries upstream of the Yakima River spawn later than most downstream populations.

The timing of upstream migration is correlated with higher flow events, such as freshets or sand bar breaches, and associated lower water temperatures. Unusual stream temperatures during spawning migration periods can alter or delay migration timing, accelerate or retard mutations, and increase fish susceptibility to diseases. The minimum stream depth necessary for successful upstream migration is seven inches (Thompson 1972, cited in Bjornn and Reiser 1991). Reiser and Bjornn (1979) indicated that steelhead preferred a depth of 9.5 inches or more. The maximum velocity, beyond which upstream migration is not likely to occur, is 8.0 cubic feet per second (Thompson 1972, cited in Bjornn and Reiser 1991).

Steelhead may spawn more than once before dying, in contrast to other species of the *Oncorhynchus* genus. It is relatively uncommon for steelhead populations north of Oregon to have repeat spawning, and more than two spawning migrations is rare. In Oregon and California, the frequency of two spawning migrations is higher, but more than two is unusual.

Steelhead spawn in cool, clear streams featuring suitable gravel size, depth, and current velocity. Intermittent streams may be used for spawning (Barnhart 1986, Everest 1973). Reiser and Bjornn (1979) found that steelhead preferred gravels of 0.5 to 4.7 inches in diameter and Smith (1973) found that flows of approximately 1.3 to 3.0 cubic feet per second were preferred by steelhead. The survival of embryos is reduced when fines of less than 0.25 inches (6.4 mm) comprise 20 to 25 percent of the substrate. Studies have shown a higher survival of embryos when intragravel velocities exceed 20 cm/hour (0.00018 ft/sec) (Phillips and Campbell 1961, Coble 1961). Steelhead eggs generally incubate between February and June (Bell 1991). The number of days required for steelhead eggs to hatch varies from about 19 days at an average temperature of 60°F to about 80 days at an average of 42°F. Fry typically emerge from the gravel two to three weeks after hatching (Barnhart 1986).

After emergence, steelhead fry usually inhabit shallow water along perennial stream banks. Older fry establish territories which they defend. Streamside vegetation and cover are essential. Steelhead juveniles are usually associated with the bottom of the stream. In smaller California streams, the water levels may drop so low during the summer that pools are the only viable rearing habitat. No passage between pools can occur until river levels rise with the onset of the rainy season. Juvenile steelhead rearing in isolated summer pools are therefore extremely vulnerable to disturbance or water quality impacts. Daytime temperatures in summer rearing pools may also be near lethal levels; riparian shading and the presence of sub-surface, cold water seeps are often essential to maintain pool temperatures at tolerable levels. In winter, juvenile steelhead become inactive and hide using any available cover, including gravel or woody debris.

Juvenile steelhead migrate little during their first summer and occupy a range of habitats featuring moderate to high velocity flows and variable depths (Bisson et al. 1988). They feed on a wide variety of aquatic and terrestrial insects, and emerging fry are sometimes preyed upon by older juveniles. Water temperatures influence the growth rate, population density, swimming ability, ability to capture and metabolize food, and ability to withstand disease of these rearing juveniles. Rearing steelhead juveniles prefer water temperatures of 45° to 58°F and have an upper lethal limit of 75°F.

Dissolved oxygen (DO) levels of 6.5 to 7.0 parts per million affected the migration and swimming performance of steelhead juveniles at all temperatures (Davis et. al. 1963). Reiser and Bjornn (1979) recommended that DO concentrations remain at or near saturation levels with temporary reductions no lower than 5.0 parts per million for successful rearing of juvenile steelhead. Low DO levels decrease the rate of metabolism, swimming speed, growth rate, food consumption rate, efficiency of food utilization, behavior, and ultimately the survival of the juveniles.

During rearing, suspended and deposited fine sediments can directly affect salmoides by abrading and clogging gills, and indirectly cause reduced feeding, avoidance reactions, destruction of food supplies, reduced egg and alevin survival, and changed rearing habitat (Reiser and Bjornn 1979). Bell (1973) found that silt loads of less than 25 parts per million permit good rearing conditions for juvenile salmoides.

Juvenile steelhead live in freshwater between one and four years (usually one to two years in the Pacific Southwest) and then become smolts and migrate to the sea from November through May with peaks in March, April, and May. The smolts can range from 5.5 to 8.0 inches in length. Steelhead spend between one and four years in the ocean (usually two years in the Pacific Southwest) (Barnhart 1986), but variations on this pattern do occur (Busby et al. 1996). Some steelhead, termed "half-pounders," return to freshwater after only a few months at sea. Half-pounders generally spend the winter in fresh water then return to sea for several months before returning to fresh water to spawn. Half-pounders occur over a relatively small geographic range in southern Oregon and northern California, including the Rogue, Klamath, Mad and Eel Rivers (Kesner and Barnhart 1972, Barnhart 1986).

Except for half-pounders, West Coast steelhead typically spend two years in the ocean before entering freshwater to spawn. The distribution of steelhead in the ocean is not well known. Coded wire tag recoveries indicate that most steelhead tend to migrate north and south along the Continental Shelf (Barnhart 1986). Steelhead stocks from the Klamath and Rogue rivers probably mix together in a nearshore ocean staging area along the northern California before they migrate upriver (Everest 1973).

Current Legal Status

Listing history

The NMFS has received numerous petitions regarding the listing of West Coast steelhead. The most comprehensive petition was submitted by the Oregon Natural Resources Council and 15 co-petitioners.(Oregon Natural Resources Council et al. 1994). In response to this petition, the NMFS established a Biological Review Team that conducted a coast-wide status review for West Coast steelhead (Busby et al. 1996). The status review identified 15 steelhead ESUs, including the Northern California ESU. Based on the results of the status review and other information, NMFS published a proposed listing determination on August 9, 1996 (61 FR 56138), proposing to list 10 ESUs in California, Oregon, Washington, and Idaho as threatened or endangered. The Northern California ESU was proposed as threatened. This ESU includes populations of steelhead between Redwood Creek in Humboldt County to the Gualala River in Sonoma and Mendocino Counties.

In a final rule published on March 19, 1998 (63 FR 13347), the NMFS determined that the Northern California ESU did not warrant listing at that time. This ESU is currently classified as a candidate species.

Threats

General threats identified in the proposed rule (61 FR 56138) include destruction and modification of habitat, overutilization for recreational purposes, and natural and human made factors. Forestry, agriculture, mining, and urbanization degrade, simplify, and fragment habitat. Water diversions and dams greatly reduce or eliminate historically accessible habitat. Sedimentation from land use activities is a primary cause of habitat degradation. The introduction of non-native species increases the level of predation. Predation by marine mammals was also identified as a concern in the proposed rule, but this was considered to be insignificant contributing factor to current declines. The NMFS (1997) has recently determined that although pinniped predation did not cause the decline of the chinook salmon populations, in localized areas where they co-occur with chinook salmon (especially where they concentrate or passage may be constricted), predation may preclude recovery. Specific areas where predation is/may preclude recovery cannot be determined without extensive studies. Natural climatic conditions may exacerbate existing habitat problems by affecting already limited spawning, rearing, and migration habitat and causing reduced ocean productivity. Hatchery programs also can threaten steelhead through competition, genetic introgression, and disease transmission.

The status review (Busby et al. 1996) specifically identified threats to the Northern California ESU. Habitat deterioration from sedimentation and flooding, apparently resulting in part from poor land management practices, was identified as a concern. Sacramento squawfish introduced in the Eel River was also noted as a possible threat. The influence of hatchery stocks, particularly in the Mad River, was considered to be a threat, both in terms of genetic introgression and of potential ecological interactions between native and introduced stocks.

Conservation needs

Although the steelhead Northern California ESU is currently a candidate species, NMFS remains concerned about the status of steelhead in this area. According to the final rule (63 FR 13347), the status of this ESU will be reevaluated in four years to determine whether listing is warranted. In the interim, the conservation needs of coastal cutthroat trout are similar to those identified for coho salmon, above.

Status and Distribution

The Northern California steelhead ESU occupies river basins from Redwood Creek in Humboldt County, California to the Gualala River, Sonoma and Mendocino Counties. Steelhead within this ESU include both winter and summer steelhead, including what is presently considered to be the southernmost population of summer steelhead, in the Middle Fork Eel River. The status review (Busby et al. 1996) noted the occurrence of half-pounder juveniles in the Mad and Eel Rivers, although Cramer et al. (1995) suggest that adults with the half-pounder life history may not spawn south of the Klamath River Basin. Several of the larger rivers in this area, including the Rogue and Klamath Rivers, are known to have migrating steelhead year-round, and seasonal runs have been identified.

Higgins et al. (1992) identified 11 summer steelhead stocks as being at risk or concern in this area. The FWS (1993) identified most stocks on U.S. Forest Service lands in this region as either depressed or critical, with only the Little Van Duzen River winter steelhead identified as stable. The status review (Busby et al. 1996) provides a detailed discussion of historical (pre-1960s) abundance of steelhead in the Northern California ESU. Based on dam counts, in the upper Eel River there was an annual average of 4,400 adult steelhead in the 1930s at Cape Horn Dam, Benbow Dam on the South Fork Eel River had an annual average of 19,000 adult steelhead in the 1940s (McEwan and Jackson 1996), and Sweasey Dam on the Mad River had an annual average of 3,800 adult steelhead in the 1940s (Murphy and Shapovalov 1951, CDFG 1994). The following discussion and description of historic abundance of the northern California ESU is excerpted from the status review (Busby et al. 1996).

In the mid-1960s, CDFG (1965) estimates of steelhead spawning populations for many rivers in this ESU totaled 198,000, broken down as follows: Redwood Creek, 10,000; Mad River 6,000; Eel River, 82,000; Mattole River, 12,000; Ten Mile River, 9,000; Noyo River, 8,000; Big River, 12,000; Navarro River, 16,000; Garcia River, 4,000; Gualala River, 16,000; and other streams (Humboldt and Mendocino Counties), 23,000. Light (1987) estimated the total run size for all major stocks in California (entire state) in the early 1980s as approximately 275,000. Of these, 22 percent were of hatchery origin, resulting in a naturally-produced run size of 215,000 steelhead for the entire state. Roughly half of this production was thought to be in the Klamath River Basin (including the Trinity River). The only current run-size estimates for this area are dam counts on the Eel River (Cape Horn Dam 115 total and 30 "wild" adults; McEwan and Jackson 1996), and summer steelhead snorkel surveys in a few tributaries that provide no total abundance estimate. Statewide adult summer steelhead abundance is estimated at about 2,000 adults (McEwan and Jackson 1996), but this estimate may refer only to early-summer steelhead entering the rivers in May, June and July, and not include the more numerous "fall-run" steelhead. While the status review (Busby et al. 1996) did not provide an overall recent estimate of abundance for this ESU, the authors interpreted the substantial declines in run sizes from historic levels at major dams in the region to be indicative of a probable overall decline in abundance from historical levels.

The status review identified two habitat blockages from dams in this ESU (Matthews Dam on the Mad River and Scott Dam on the Eel River, McEwan and Jackson 1996), and speculated that other minor blockages (impassable culverts, etc.) were likely throughout the region. Habitats throughout the northern coast of California were severely impacted by catastrophic flooding in 1964, and damages from this flooding were likely exacerbated by poor land us practices prior to this event (McEwan and Jackson 1996).

Forest practices have also contributed to the incremental degradation of stream habitats (Higgins et al. 1992, McEwan and Jackson 1996) and excessive sedimentation and unstable spawning gravels have been cited as major habitat problems (CDFG 1991, Higgins et al. 1992).

A high abundance of non-native Sacramento squawfish (*Ptychocheilus grandis*) have been reported recently in the Eel River Basin (Brown and Moyle 1991, Moyle and Yoshiyama 1992), suggesting increased risks of predation on juvenile steelhead.

The status review provides a computation of trends for seven stocks within the northern California steelhead ESU based on adult escapement information. Of these seven, five stocks were found to exhibit declines and two stocks suggested increase over the available data series, (ranging from a 5.8 percent annual decline to a 3.5 percent annual increase). Three (all decreasing) of these trends were significantly different from zero (see Appendix C, status review). Analysis of one long data set representing counts from the Eel River and Cape Horn Dams, suggested that the major, and more significant, stock declines occurred prior to 1970.

Hatchery fish are widespread within this ESU and are reported to spawn naturally throughout the region. Past and present hatchery practices present a major threat to the genetic integrity for steelhead in this ESU. The status review cites McEwan and Jackson (1996), as concluding "despite the large number of hatchery smolts released, steelhead runs in north coast drainages are comprised mostly of naturally produces fish." However, very little information was available to the BRT on the actual contribution of hatchery fish to natural spawning, or present total run sizes for this ESU.

The primary steelhead hatchery within the range of this ESU is Mad River Hatchery, established in 1971 by CDFG for fishery enhancement reasons (McEwan and Jackson 1996). The Mad River Hatchery winter steelhead stock was founded with steelhead eggs from the Eel River and the San Lorenzo River and is reviewed by Cramer et al (1995). Eel River steelhead eggs are still being transferred to the Mad River hatchery for rearing and subsequent release back into the Eel River. CDFG (1994) estimates that approximately 233,000 juvenile steelhead of various stock origins are released annually into Mad River. Cramer et al. (1995) estimates that all other basins in this area together receive about 75,000 steelhead per year, for a total annual hatchery release of at least 404,000 steelhead within the range of the northern California steelhead ESU (Busby et al. 1996)

<u>**Southern Oregon/California Coasts ESU coastal cutthroat trout**</u>

Species description

The coastal cutthroat trout is one of 14 extant subspecies of cutthroat trout distributed throughout western North America. The cutthroat trout was historically one of the most broadly distributed salmoides in western North America. They were found in many lakes and streams throughout the coastal and interior American west as far east as the Rio Grande drainage (Rio Grande cutthroat trout) and the eastern slope of the Rocky Mountains (Yellowstone and greenback cutthroat trout). Approximately one million years ago, cutthroat trout are believed to have diverged into a coastal group with 68 or 70 chromosomes and an interior group (Westslope cutthroat trout) with a 66 chromosome set. While the interior group continued to diverge into a 64 chromosome set (Lahontan and Yellowstone cutthroat trout, and more recent “minor” subspecies), the coastal cutthroat trout has remained essentially intact. Further discussion about the evolution and systematics of cutthroat trout can be found in Behnke (1992).

Coastal cutthroat trout differ from other cutthroat trout not only in its chromosome set but in its diverse life history. It is the only cutthroat subspecies with an anadromous form. Coastal cutthroat trout range from Prince William Sound, Alaska, south to the lower Eel River, California. Inland, the subspecies' range rarely extends farther than 100 miles and is usually less than 62 miles. In California, Oregon, and Washington, eastern distribution is limited by the Cascade Mountains. Coastal cutthroat trout belong to the same genus as Pacific salmon and steelhead, but are generally smaller. In freshwater, coastal cutthroat can be distinguished by a bright red slash under the jaw and dark, irregular spots over the entire body. Coastal cutthroat trout are believed to have the healthiest populations of all the cutthroat trout, as they have experienced the least habitat destruction, hybridization, or over fishing (reviewed in Pauley et al. 1989, Trotter 1989, and Trotter et al. 1993). Nevertheless, the AFS Endangered Species Committee identified all populations of anadromous cutthroat trout as being at some risk of extinction (Nehlsen et al. 1991).

Life History

General life history information on coastal cutthroat trout is summarized below. Further information on this species can be found in Hall et al. (1997), and the petition to list sea-run (coastal) cutthroat trout (Oregon Natural Resources Council et al. 1997). Information for this discussion was also taken from the Status review of Coastal Cutthroat Trout from Washington, Oregon, and California" (Johnson et al. 1999).

The life history of coastal cutthroat trout may be the most complex of any Pacific salmonid (Northcote 1997). Coastal cutthroat trout exhibit a range in the timing of life-history events such as migrations and spawning. Furthermore, coastal cutthroat trout are mostly iteroparous, spawning three to five times during their life cycle. Finally, many populations have both migratory and nonmigratory fluvial and lacustrine populations as well as anadromous forms. All populations with access to the ocean are believed to have anadromous members, but not all members migrate to the sea every year (reviewed in Trotter 1997).

Johnson et al. (1999) identified four life history forms based on migratory behavior: anadromous (sea-run or coastal-type), resident (non-migratory or "cascade"), potamodromous (river-migrating or fluvial), and adfluvial (lake-dwelling or lacustrine). The anadromous form migrates from freshwater spawning areas to feed during the summer in marine environments, returning to fresh water in the winter to feed, seek refuge, or spawn, typically returning to the ocean the next spring. The resident form is usually found in upper headwater tributaries and is generally considered to be nonmigratory and to maintain small home territories. Resident cutthroat trout apparently grows more slowly than other forms, are smaller at maturity, and rarely live longer than 2-3 years (Wyatt 1959; Nicholas 1978; June 1981). The potamodromous form migrates within large river basins, but does not move out to sea. This form is rarely found below barriers or in locations with access to anadromous fish. The adfluvial form lives and feeds within lake systems and may spawn either in the shallows of the lake or migrate up tributaries to suitable spawning habitat.

The distinction between these life-history forms can be ambiguous. Adfluvial fish may also exhibit anadromy if they have access to the sea (Armstrong 1971). Fish considered to be resident in one year may migrate to the ocean the next year (Sumner 1962; Giger 1972). Conversely, anadromous fish do not necessarily return to the ocean after spawning, but may remain in freshwater for a year (Giger 1972; Tomasson 1978). The distinctions between forms can be further blurred where multiple forms are found in sympatry, because they are often morphologically indistinguishable, particularly as juveniles (Fuss 1982).

Regardless of life history form, coastal cutthroat trout spawning typically starts in December and continues through June, with peak spawning in February (reviewed in Pauley et al. 1989; Trotter 1989). In California, spawning begins in November, with peak spawning in late December in the larger river basins and late January and February in the smaller coastal rivers and streams (reviewed in Johnson 1999). Spawning occurs in small tributaries with low gradients and low summer flows, usually less than 10 cubic feet per second. Redds are primarily built in the tails of pools (Johnston 1982), in gravel sized from 0.2 to 2 inches (pea to walnut-size) (Cramer 1940), and at depths of 6 to18 inches (Jones 1978). Generally, coastal cutthroat trout spawn upstream of coho salmon or steelhead. It is believed that this choice of spawning sites has evolved to reduce competitive interactions between young-of-the-year cutthroat trout and other salmoides (Johnson et al. 1999). This may be important as coastal cutthroat trout typically emerge later and are smaller in size than other salmonid species.

Coastal cutthroat trout are iteroparous and have been documented to spawn each year for at least five years, although some fish do not spawn every year and some do not return to seawater after spawning (Giger 1972). Spawners may experience high post-spawning mortality due to weight loss (Sumner 1953) and other factors (Cramer 1940, Sumner 1953). Still, Sumner (1953) observed in one Oregon stream that over 39 percent of one year's spawning population returned to spawn the following year, 17 percent returned for a third year, and 12 percent returned for a fourth year. In contrast, in another Oregon stream that had an intense sport fishery, Giger (1972) noted that only 14 percent of spawners returned to spawn the following year. This heavy mortality of first year spawners may have implications for the health of these populations, as the fecundity of female coastal cutthroat trout increases with age. The eggs of older females, those that have spawned two or three times, are more numerous and larger than the eggs of first-time spawners (cited in Trotter 1997). Larger eggs develop into larger alevins, which have size advantages in subsequent growth and survival (reviewed in Trotter 1997).

Coastal cutthroat trout eggs hatch within six to seven weeks after spawning. Citing several studies, Trotter (1997) reported that the eggs of coastal cutthroat trout require between 362 and 500 degree-days to hatch (expressed in units of days above a threshold temperature of 32°F). Depending on water temperatures, alevins emerge between March and June, with peak emergence in mid-April (Giger 1972, Scott and Crossman 1973). Fry quickly move to channel margins and backwaters.

The literature is conflicting over habitat preferences for juvenile coastal cutthroat trout. This disagreement in the literature has been explained by some researchers as evidence that habitat preference by coastal cutthroat trout is affected by inter-species competition. When they are the only salmonid in the stream, Glova (1984) found that juvenile coastal cutthroat trout were more abundant in pools, but use riffles and glides as well. When coho salmon fry and sculpins are present, the coastal cutthroat trout juveniles are more evenly distributed (Glova 1987). Studies of summer rearing habitat preferences found that where coho salmon are also present, the majority of coastal cutthroat trout found in pools were age one or older. June (1981), Bisson and Sedel (1984), and Mitchell (1988) report that juvenile coastal cutthroat trout prefer shallower, swifter-water habitats such as low-gradient riffles and pool tailouts. Glova and Mason (1976, cited in Trotter 1997), in contrast, reports that in allopatric situations, coastal cutthroat trout young-of-the-year select pools and other slow water habitats. Where juvenile steelhead are also present, they also dominate the young-of-the-year cutthroat trout in riffles (reviewed in Trotter 1997). Older (age 1+) coastal cutthroat trout select pools, particularly those with root wads or other LWD (Bisson and Sedel 1984). Water temperatures are probably a factor in habitat selection; coastal cutthroat trout do not fare well when water temperatures exceed 72°F (Pauley et al. 1989).

The segregation between species is not limited to habitat; food preferences also differ between species. Glova (1984) observed that coho salmon fry ate more adult insects, Dipthera and Hemiptera, while coastal cutthroat trout ate more larvae and pupae, mostly chironomids. The author further suggested that coastal cutthroat trout were more generalists, utilizing both benthic and drift, while salmon ate only drift. Martin (1984, cited in Trotter 1997) also noted that coastal cutthroat trout were generalists; in a study spanning several months (February through November), the author observed that all age classes of coastal cutthroat trout ate the same small items throughout the study, but that prey shifted from aquatic to terrestrial organisms as these became more abundant in the autumn.

The territoriality between species decreases in the winter. Deep pools, particularly those with log jams, root wads, and overhanging banks, are preferred (Bustard and Narver 1975) by coastal cutthroat trout, along with pools with cobble and boulder substrate. Off-channel habitats are used by coastal cutthroat trout during winter months, as are lakes, if they are present in the drainage and accessible to cutthroat trout (Armstrong 1971). Coastal cutthroat trout usually remain in upper tributaries of watersheds until they are one year of age, when they may begin moving more extensively throughout the river system. Giger (1972) observed that downstream movement by juveniles in the Alsea River, Oregon, began with the first spring rains, usually mid-April with peak movement in mid-May. Giger (1972) also noted that many of these fish did not smolt, rather they remained in the estuary over the summer. The first winter rains apparently triggers an opposite migration. In Oregon, Washington, and British Columbia, the upstream movement of juveniles with parr marks (indicating they had not undergone smoltification) from estuaries and mainstems up into the tributaries of river systems begins with the onset of winter freshets during November (Giger 1972, Moring and Lantz 1975, Cederholm and Scarlett 1982, Hartman and Brown 1987, and Garrett 1998), and continues through the spring, frequently peaking during late winter and early spring (Cederholm and Scarlett 1982, Hartman and Brown 1987, Garrett 1998).

Differentiating between juveniles and smolts is more difficult with coastal cutthroat trout than with other Pacific salmoides. Because coastal cutthroat trout migrate extensively within river systems it is difficult to determine which fish are seaward bound and which are simply moving from one freshwater location to another. In general, for anadromous cutthroat trout, smoltification occurs after two to four years in freshwater (Sumner 1962, Lowry 1965, Giger 1972, Michael 1980. Fuss 1982). In Alaska, the initial saltwater entrance occurs at between four and six years of age (Armstrong 1971, Jones 1978). Saltwater entry typically occurs between March and July, but varies depending on location. In California, smolt emigration begins in March and continues through June and July, with peak migration occurring in April and May (Redwood National Park 1983, 1988-1993, 1997, 1998, unpublished data, Brown 1988, Shaw and Jackson 1994, Simondet 1997, Voight and Hayden 1997). This is about a month earlier than spawned-out adults that are returning to salt water. In Oregon and Washington, spawned-out adults will return to salt water beginning in late March through early April (Trotter 1997).

Once in the ocean, coastal cutthroat trout remain in nearshore waters and do not stay in salt water for very long. Studies by Giger (1972) and Jones (1973, 1974, 1975) indicated that coastal cutthroat trout, whether first time or seasoned migrants, stayed close to shore, rarely crossed bodies of water larger than 5.6 miles wide, and remained at sea for an average of 91 days, with a range of 5-158 days. Some studies indicate that coastal cutthroat trout prefer areas with freshwater influence while in saltwater (Loch and Miller 1988, Pearcy et al. 1990).

Most anadromous coastal cutthroat trout return to freshwater for winter, but not necessarily to spawn. Sexually immature anadromous coastal cutthroat trout will overwinter in freshwater, return to saltwater the following summer to forage, then return to freshwater the following winter to spawn (Trotter 1997). In Alaska, the percentage of sexually immature fish returning to freshwater was nearly 50 percent (Jones 1972-1976), but in Oregon, Sumner (1953) found that about 95 percent of migrants, including first-time migrants, were sexually mature. It has been suggested that non-spawning migrants are more likely to wander during their first return to freshwater than sexually mature fish. Jones (1975, 1976) reported that tagged fish from Petersburg Creek in southeastern Alaska were captured in 13 nearby streams the following year, but during the second year, a much higher portion of tagged fish were captured back in Petersburg Creek. Johnston (1982) also suggested that sexually immature first-year migrants may conduct feeding runs to non-natal rivers, but Johnston also proposed that these fish would return to their natal stream the following year, when they were sexually mature.

The timing of freshwater entry varies widely throughout the species range, from late June through the following April. Some river systems appear to have two distinct freshwater entry times (Sumner 1972, Johnston 1982). Sumner (1972) noted that in several Oregon rivers that had well-developed estuary systems, some coastal cutthroat trout entered the estuary in July and remained there through the rest of the summer before moving upriver with the first fall freshets. A second group of fish entered the same rivers in the fall, moving immediately upstream. In California, when there is year-round access to the ocean, coastal cutthroat trout immigration beginning in late July and continuing through December, with peak migration in September and October (CDFG annual seining results, unpublished data 1980-1989, USFS annual observation counts, unpublished data 1992-1997). In smaller rivers where seasonal sand bars block access, adult immigration begins with the opening of the sand bar, usually with the first winter freshets in November or December, and continues through March, with peak migration occurring in January and February (Redwood National Park annual reports, unpublished data 1983, 1988-1993).

Current Legal Status

Listing history

The NMFS originally initiated a review of the status of coastal cutthroat trout after receiving a petition from the Oregon Natural Resources Council, the Wilderness Society, and the Umpqua Valley Audubon Society (Oregon Natural Resource Council et al. 1993) to list the North and South Umpqua River sea-run cutthroat trout as threatened or endangered. The status review concluded that the Umpqua River coastal cutthroat trout did constitute an ESU (Johnson et al. 1994), and NMFS subsequently listed the Umpqua River coastal cutthroat trout as an endangered species on August 9, 1996 (61 FR 41514).

In response to a petition received by NMFS from the Oregon Natural Resources Council and others on December 5, 1997, to "list the sea-run cutthroat trout as threatened or endangered throughout its range in the states of California, Oregon, and Washington" (Oregon Natural Resources Council et al. 1997), the NMFS recently conducted a status review on coastal cutthroat trout throughout the Pacific Northwest (Johnson et al. 1999). The draft status review has identified six coastal cutthroat trout ESUs, including the (SOCC) ESU.

Threats

General threats identified in the draft status review (Johnson et al. 1999) include oceanic and climatic changes, threats to genetic integrity such as artificial propagation of coastal cutthroat trout and hybridization between coastal cutthroat trout and steelhead, and recreational fisheries. Within the SOCC ESU, the draft status review noted that severe habitat degradation, due to logging, road construction, and some local development, has contributed to a reduction in habitat capacity relative to historical population levels. Water withdrawals have also affected the size and habitat quality of some estuaries in larger river basins. The risks due to interactions with hatchery cutthroat trout are considered to be low in this ESU, but interactions with naturally occurring or hatchery produced coho salmon and steelhead are considered deleterious. The incidental capture of coastal cutthroat trout in sport fisheries targeting steelhead and coho salmon was also noted as a possible threat to coastal cutthroat trout in the SOCC ESU.

Conservation needs

The status review for the coastal cutthroat trout SOCC ESU states that although populations within this ESU are below historical levels, this ESU is not presently endanger of extinction, nor is it likely to become so in the foreseeable future. In general, the conservation needs of coastal cutthroat trout are identical to those identified for coho salmon, above.

Status and Distribution

Available historical and recent coastal cutthroat trout abundance information is summarized in the status review for this species (Johnson et al. 1999). The following discussion is taken from that document and from Hall et al. (1997).

In California, coastal cutthroat trout have been observed in 182 named streams, about 71 percent of which are within the species' California range. Reproducing populations occur throughout most of the Humboldt Bay tributaries, the Smith and Little Rivers, the lower portions of Redwood Creek, Klamath, Mad, and Eel Rivers, and numerous small named and unnamed coastal tributaries (Gerstung 1997). Coastal cutthroat trout also occur in five coastal lagoons and ponds - Big, Stone, and Espa lagoons, and the Lake Earl-Talawa complex (Gerstung 1997). In California, almost 46 percent of coastal cutthroat trout occupied habitats in the Smith and Klamath River drainages. Electroshocking data from a number of California streams suggest that densities of juvenile or smolting cutthroat trout (50-200mm) are consistently between 15 and 30 fish per kilometer of stream below barriers to anadromy (Gerstung 1997). Densities of coastal cutthroat trout in the same size range above the barriers were much higher, ranging from 60-400 fish per kilometer. Adult densities below barriers in the Smith River were 3-12 cutthroat trout per kilometer; 6-12 adults per kilometer were found in the Little River. In contrast, researchers in Redwood Creek only found 0.5 adults per kilometer (Gerstung 1997).

Trends in smolt abundance from Mill Creek, a tributary to the Smith River, have been increasing since 1994 (Howard and Albro 1995, 1997). Estimated smolt numbers in this tributary have ranged from 750 to nearly 4,000 fish over a four-year sampling period. Data from the lower Klamath River between 1980 and 1991 showed a variable but relatively stable trend in the number of coastal cutthroat trout caught per seine haul (Gerstung 1997). In contrast, data from the Klamath River estuary indicated a dramatic increase in the mean number of cutthroat trout caught per minute between 1991 and 1995 (Gerstung 1997).

In Oregon, coastal cutthroat trout are widespread west of the Cascade Mountain crest. All life-history forms - resident, fluvial, adfluvial, and anadromous - are present in coastal streams within the SOCC ESU. Trends in smolt abundance in the Winchuck River have been declining over the past three years; from 1996 to 1998, 2,800, 1,990, and 1,400 smolts have been trapped (T. Confer, Oregon Department of Fish and Wildlife, unpublished data). Coastal cutthroat trout sampled in a downstream weir on Jack Creek, a tributary to the Chetco River, were stable in numbers from 1989-91, ranging from 643-667 total cutthroat trout (T. Confer, Oregon Department of Fish and Wildlife, unpublished data). Historically, the Rogue River had a substantial recreational cutthroat trout fishery, suggesting a high abundance of coastal cutthroat trout in that basin. There is only one known recent estimate of smolt abundance in the Rogue

River: from March-May, 1998 a downstream migrant trap on Lobster Creek, a tributary to the lower Rogue River, caught 146 smolts, 66 juveniles, and four adult cutthroat trout (T. Confer, Oregon Department of Fish and Wildlife, unpublished data). Correcting for trap efficiency, ODFW biologists estimated that 838 cutthroat trout smolts migrated past the trap.

Over the whole SOCC ESU, coastal cutthroat trout are thought to be widely distributed in many small populations. Two possible exceptions are populations in the Rogue and Smith River basins, where the abundance of cutthroat trout may be comparatively high, although the lack of information on smolt-to-adult survival makes interpreting smolt abundance estimates in these rivers difficult. Population sizes in other streams throughout this ESU are thought to be small, in part because it is the southern limit of the coastal cutthroat trout subspecies. The draft status review notes that severe habitat degradation has occurred in this region due to logging and some local development, which probably have contributed to reduction in habitat capacity. In addition, seasonal dewatering of stream mouths naturally occurs in northern California, but the extent to which human activities have altered the natural cycles of river mouth blockages in this area is unknown.

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